FINAL REPORT

Estimated Exposure and Lifetime Cancer Incidence Risk from Plutonium Released from the 1957 Fire at the Rocky Flats Plant

Part of Task 3: Independent Analysis of Exposure, Dose, and Health Risk to Offsite Individuals

August 1999

Submitted to the Colorado Department of Public Health and Environment, Disease Control and Environmental Epidemiology Division, Rocky Flats Health Studies in partial fulfillment of contract No. 100APPRCODE 391

"Setting the standard in environmental health"

RAC

Radiological Assessments Corporation
417 Till Road Neeses, South Carolina 29107
phone 803.536.4883 fax 803.534.1995

FINAL REPORT

Estimated Exposure and Lifetime Cancer Incidence Risk from Plutonium Releases from the 1957 Fire at the Rocky Flats Plant

Part of Task 3: Independent Analysis of Exposure, Dose, and Health Risk to Offsite Individuals

August 1999

Authors

Arthur S. Rood, K-Spar, Inc.
Helen A. Grogan, Ph.D., Cascade Scientific Inc.

Principal Investigator

John E Till, Ph.D., *Radiological Assessments Corporation*

EXECUTIVE SUMMARY

The Rocky Flats Environmental Technology Site is owned by the U.S. Department of Energy (DOE) and is currently contractor-operated by Kaiser-Hill Company. For most of its history, the site was called the Rocky Flats Plant (RFP) and was operated by Dow Chemical Company as a nuclear weapons research, development, and production complex. The RFP is located about 8–10 km (5–6 mi) from the cities of Arvada, Westminster, and Broomfield, Colorado, and 26 km (16 mi) northwest of downtown Denver, Colorado.

Through a 1989 Agreement in Principle between DOE and the State of Colorado, DOE provided the State with funding and technical support for health-related studies. The purpose of the Historical Public Exposures Studies on Rocky Flats is to estimate exposure to nearby residents from past toxic and radioactive releases from the plant.

This report documents fate and transport calculations and lifetime cancer incidence risk estimates for inhalation of plutonium[1] released during the glove box fire that occurred on September 11, 1957. Risk estimates are reported in terms of probability distributions that reflect the uncertainty in the calculation. It presents estimates of time-averaged plutonium airborne concentrations at selected receptor locations within the model domain. Lifetime cancer incidence risks are then calculated for hypothetical individuals residing in the model domain who inhale plutonium. Behavior and physical attributes of hypothetical individuals are characterized by exposure scenarios that are discussed in detail. Details of atmospheric transport modeling and uncertainty estimates are described. Atmospheric transport calculations and risk estimates made in Phase I are summarized, and an overview of the source term developed for Phase II and documented in Voillequé (1999a) is provided. Lifetime cancer incidence risks using risk coefficients and associated uncertainty developed by Grogan et al. (1999) are summarized.

Summary of the Glove Box Fire. The fire in Building 771 began in a glove box in Room 180 at 10:06 p.m. September 11, 1957, probably as a result of spontaneous ignition of metallic plutonium casting residues. The primary fire in Room 180 was extinguished by 10:38 p.m.; however, the fire spread to the exhaust filter plenum system, culminating in an explosion in the ventilation system at 10:39 p.m. Brief flareups of the fire in Room 180 after the explosion were promptly put out. By 2 a.m., water had been applied to the plenum filter fire for more than 2 hours. Analysis, based upon post-fire experimental evidence, indicates that the flammable Chemical Warfare Service (CWS) filters used in various effluent treatment systems burn rapidly. It is estimated that the largest releases occurred between 10:15 p.m. and 10:45 p.m., before the primary fire was extinguished and before water was applied to the plenum filter fire. Exhaust fans were off at 10:40 p.m. when the fire burned through the power cable. After 10:45 p.m., the primary fire was out, the exhaust fans were off, and releases were much lower. After that time, the principal contributor to the releases was the burning filter plenum.

Review of Phase I Evaluation of the 1957 Fire. In Phase I of the historical public exposures study, the 1957 fire was determined to be the second largest plutonium release event at Rocky Flats. To analyze the 1957 fire, ChemRisk divided it into four main time periods:

[1] In this context, the word plutonium means weapons grade plutonium, which consists primarily of ^{239}Pu (≈ 93.8%), ^{240}Pu (≈5.8%), and ^{241}Pu (≈0.36%) by weight percent. Specific activity of weapons grade plutonium is 0.072 Ci g^{-1}.

1. From the start of the fire until the explosion
2. The explosion
3. After the explosion up to the time that the filter fire was controlled
4. From the time the filter fire was controlled until the last fire was extinguished.

Based on these four time periods, a two-stage release was evaluated. The first stage was of short duration and included the time from the start of the fire through the filter plenum explosion. This release of plutonium was characterized by coarse particles that deposited within a relatively short distance, and it did not contribute to offsite exposures. The second stage was of longer duration and was characterized by the release of fine particles during the remainder of the fire.

ChemRisk used the INtegrated PUFF dispersion code version 2 (INPUFF2) to model the atmospheric dispersion and deposition of the released plutonium. Air dispersion calculations that were calibrated to vegetation data estimated a coarse particle release of 60 μCi, with an upper and lower bounds of the 95% confidence interval of 1900 μCi and 1.9 μCi, respectively. The best estimate of the second stage fine particle releases was 0.07 Ci, with the upper and lower bounds of the 95% confidence interval of 2.4 Ci and 0.002 Ci, respectively. Maximum effective dose equivalent for inhalation of plutonium in the model domain was estimated to have a geometric mean of 1.8×10^{-5} Sv, with a geometric standard deviation of 6.2. Cancer risk estimates were made using a conversion of 7.3% Sv^{-1}. The maximum cancer risk was 1×10^{-6}, with a lower and upper bound 95% confidence interval of 3×10^{-8} to 5×10^{-5}, respectively.

Phase II Release Estimates for the 1957 Fire. Phase II release estimates were substantially higher than those for Phase I. Total release quantities ranged from 11 Ci at the 5% level to 36 Ci at the 95% level. Distributions of release quantities were reported in terms of percentiles in 5% increments for each 15-minute period of the event. The 15-minute interval was chosen to match the resolution of the meteorological data and, thereby, facilitate the calculations of environmental transport and risk. The fire event was modeled beginning at 10:00 p.m. on the evening of September 11, 1957, and ending at 2:00 a.m. the morning of September 12, 1957. Most of the release was postulated to occur during the first hour of the accident as a result of the explosion in the ventilation system and filter fire.

The sizes of plutonium particles that were released during the fire are not known. However, release estimates assumed all activity was respirable and could have ranged from submicron to up to 10 μm. For this reason, particle size was treated stochastically in the fate and transport calculations.

The release estimates made for Phase II are considerably larger than for those of Phase I, which relied on environmental monitoring data in vegetation to estimate source terms.

Environmental Transport Modeling. Five atmospheric transport models, ranging from a simple straight-line Gaussian plume model to a complex terrain model, were evaluated for use in this study (Rood 1999a). Models were compared to tracer measurements taken in the winter of 1991 at Rocky Flats. The results of this evaluation indicated no one model clearly outperformed the others. However, the puff trajectory models (RATCHET, TRIAD, and INPUFF2) generally had lower variability and higher correlation to observed values compared to the other models. The RATCHET model was chosen for these calculations because it incorporates spatially varying meteorological and environmental parameters. Additionally, the model includes modules that

perform random sampling of the meteorological parameters, allowing for Monte Carlo analysis of uncertainty.

The model domain encompassed a 2200-km^2 area (50 km north-south × 44 km east-west). The domain extended 28 km south, 12 km west, 22 km north, and 32 km east from the RFP. Most of the Denver metropolitan area and the city of Boulder were included in the domain.

Meteorological data from RFP for the time of the fire were limited to the information in the letter to Dr. Roy Cleare, Executive Director of the Colorado Department of Health, dated March 20, 1970. These data were supplemented with data from Denver Stapleton International Airport. Hourly stability classes were calculated separately for the RFP and Denver Stapleton International Airport meteorological recording stations using the general classification scheme discussed in Pasquill (1961), Gifford (1961), and Turner (1964).

The RATCHET simulation considered plume rise from both momentum and buoyant forces. Burning of the filter plenum resulted in effluent release temperatures near 400°C and effluent velocity of 6.37 $m^3\ s^{-1}$. Plume rise calculations indicated a final plume height to range from 45 to 75 m above the 44-m stack. Model simulations were performed using the smaller of these two values.

Treatment of Uncertainty. Risk estimates were reported as probability distributions that reflect our current state of knowledge of the problem. They do not represent the probability of a seeing a health effect within the population of potential receptors. Uncertainty estimates for atmospheric transport modeling employed the Monte Carlo sampling features of the RATCHET code, which considered uncertainty in the wind speed, wind direction, Monin-Obukhov scaling length, and mixing height. This allowed for mass balance of material within the model domain for each of the 1000 Monte Carlo trials performed. Monte Carlo techniques were used to propagate atmospheric model prediction uncertainty through to the final risk calculations.

Predicted Air Concentrations. Distributions of 9-hour time-integrated concentrations (*TICs*) for 15 receptor locations were described in terms of the 5th, 50th, and 95th percentile values of the distribution of predicted concentration values. These statistics were used to describe the concentration distributions and were not used in the risk calculation. The actual distributions comprising 1000 RATCHET realizations were used instead to calculate plutonium intake and risk to the receptors. The 15 receptor locations chosen for risk calculations represented individuals from each of the major population centers in addition to receptors placed at locations of high concentration in the model domain. Values of *TIC* in the model domain ranged from minimum of 0 to a maximum of 14 pCi-h m^{-3} at the 5% level and 0 to 3600 pCi-h m^{-3} at the 95% level. Cities where 95% of the 1000 Monte Carlo trials had a concentration greater than zero included Arvada, Westminster, Federal Heights, Thornton, and Northglenn. Unlike the routine release evaluation (Rood 1999b) where nonzero concentrations were calculated for all receptor nodes, this event has an additional component of uncertainty, that is, the probability that the plume even reaches the receptor.

Exposure Scenarios. The risk that a person experiences depends upon a number of factors, such as

- Where the person lived and worked in relation to the RFP
- Did the person live near the RFP during the 1957 fire

- Lifestyle (that is, did the person spend a great deal of time outdoors or doing heavy work on a farm)
- Age and gender of the person.

To consider these features of a person's life, we developed profiles (or exposure scenarios) of hypothetical, but realistic, residents of the RFP area for which representative risk estimates could be made. Risks were calculated for seven hypothetical exposure scenarios. These scenarios incorporate typical lifestyles, ages, genders, and lengths of time in the area. They can help individuals determine risk ranges for themselves by finding a lifestyle profile that most closely matches their background. The scenarios were not designed to include all conceivable lifestyles of residents who lived in this region during the time of RFP operations. Rather, they provide a range of potential profiles of people in the area.

The seven exposure scenarios were distributed at 15 locations within the model domain. Receptors were placed in major population centers and at points where the maximum concentrations in the model domain were observed. Scenarios included a rancher, housewife, infant, child, student, and laborer.

We only considered the inhalation pathway in this evaluation. We made this decision based on Phase I results that showed soil ingestion and inhalation of resuspended plutonium were minor pathways when considering the long-term exposure to Rocky Flats effluent (ChemRisk 1994a). We recognize that these two later pathways become increasingly important for the later years of exposure because of the accumulation of deposited plutonium in soil and the lower airborne emissions. However, doses during this period (1971–1989) were several orders of magnitude smaller than doses for earlier years (1952–1970). The inhalation of resuspended plutonium will be addressed later in a comprehensive risk report covering all releases from the RFP.

Each receptor scenario incorporates inhalation rates that reflect the receptor's lifestyle and the fact that the fire occurred in the evening when most people would be preparing for bed. However, a scenario where a laborer was working the graveyard shift was also included. Uncertainty was not incorporated into the exposure scenarios; that is, the physical attributes and behavior of the receptors were assumed to be fixed. The calculated risks are not intended to represent a population of receptors who exhibit differing behaviors.

Plutonium Risk Coefficients. Lifetime cancer incidence risk coefficients (risk per unit intake) with uncertainty for plutonium were developed by Grogan et al. (1999) for the four critical organs: lung, liver, bone surface, and bone marrow (leukemia). Where feasible, gender- and age-specific risk coefficients were determined. Risk coefficients were reported for three different particle size distributions having geometric mean values of 1 μm, 5 μm, and 10 μm activity median aerodynamic diameter and a geometric standard deviation of 2.5 in all cases.

Incremental Lifetime Cancer Incidence Risk. Incremental lifetime cancer incidence risks were expressed in terms of percentiles of the cumulative density function. The value of the incremental lifetime cancer incidence risk depended on not only the location of the receptor but also the percentile value chosen. At the 2.5% level, the receptor with the maximum total (all organs) risk in the model domain was the laborer located east of the city of Arvada (5.8×10^{-7}), followed by the laborer located at Federal Heights (5.4×10^{-7}). At the 50% level, the receptor with the maximum risk was also the laborer located east of Arvada, but the receptor with the

second highest risk was the rancher located south of the RFP buffer zone. At the 97.5% level, the rancher located south of the RFP buffer zone had the highest risk (1.7×10^{-4}), followed by the rancher located west of Arvada (3.4×10^{-5}). The 95% uncertainty range (from 2.5% to 97.5%) for the rancher located south of the RFP buffer was from 0 to 1.7×10^{-4}. Using the laborer located east of Arvada as an example, the uncertainty in these risk estimates may be interpreted as follows:

- There is a 95% probability that the incremental lifetime cancer incidence risk was between 5.8×10^{-7} (2.5% value) and 1.6×10^{-5} (97.5% value).
- There is a 2.5% probability that the incremental lifetime cancer incidence risk was greater than 1.6×10^{-5} and a 2.5% probability the risk was less than 5.8×10^{-7}.

We can also interpret this to mean that given an exposure history and lifestyle similar to the laborer, there is a 97.5% probability that the model predicted number of cancer cases attributed to inhalation of plutonium originating from the 1957 fire release would be no greater than 16 persons in a population of 1 million similarly exposed individuals. The organ with the greatest risk was the lung, followed by the liver, bone, and bone marrow.

Receptors located in the cities of Arvada, Westminster, Northglenn, Thornton, and Federal Heights had the highest probability of being exposed. In the 1000 RATCHET Monte Carlo trials, the plume never reached the city of Boulder; therefore, the risk was zero at that location. While the rancher located south of the RFP buffer zone had the highest risk at the 97.5% level, the probability of him being exposed was only about 60% (that is, only 60% of the RATCHET simulations predicted the plume to reach that location). This added level of uncertainty makes it difficult to interpret results in the same manner as those for routine releases.

An almost infinite number of possible exposure scenarios can be defined; in most cases, the risks associated with each scenario will differ. However, the maximum risks will probably be bounded by the risks associated with the rancher and laborer scenarios. These scenarios may be considered the maximum exposed individual in the model domain because they were placed at the point of highest concentration outside the RFP buffer zone. In addition, the laborer was assumed to be working a graveyard shift, thereby maximizing his breathing rate during the releases from the fire. The calculated risks were within the U.S. Environmental Protection Agency point of departure for acceptable lifetime cancer incidence risk of 1 in 1,000,000 to 1 in 10,000 people.

CONTENTS

FIGURES

TABLES

ACRONYMS

AED	aerodynamic equivalent diameter
AMAD	activity median aerodynamic diameter
ASCOT	Atmospheric Studies in Complex Terrain
CDPHE	Colorado Department of Public Health and Environment
CWS	Chemical Warfare Service
DOE	U.S. Department of Energy
EPA	U.S. Environmental Protection Agency
GM	geometric mean
GSD	geometric standard deviation
HAP	Health Advisory Panel
HEPA	high-efficiency particulate air (filter)
ISC	Industrial Source Complex Short Term Version 2
INPUFF2	INtegrated PUFF dispersion code, version 2
LET	linear energy transfer
MPL	maximum permissible level
RAC	*Radiological Assessments Corporation*
RATCHET	Regional Atmospheric Transport Code for Hanford Emission Tracking
RBE	relative biological effectiveness
RFP	Rocky Flats Plant
TIC	time-integrated concentration
TLLa	total long-lived alpha activity
TRAC	Terrain Responsive Atmospheric Code
USGS	U.S. Geological Survey
UTM	universal transverse mercator
WVTS	Winter Validation Tracer Study

INTRODUCTION

The Rocky Flats Environmental Technology Site is owned by the U.S. Department of Energy (DOE) and is currently contractor-operated by Kaiser-Hill Company. For most of its history, the site was called the Rocky Flats Plant (RFP) and was operated by Dow Chemical Company as a nuclear weapons research, development, and production complex (Figure 1). The RFP is located on approximately 2650 ha (6500 acres) of Federal property, about 8–10 km (5–6 mi) from the cities of Arvada, Westminster, and Broomfield, Colorado, and 26 km (16 mi) northwest of downtown Denver, Colorado. The original 156-ha (385-acre) main production area is surrounded by a 2490-ha (6150-acre) buffer zone that now delineates the RFP boundary.

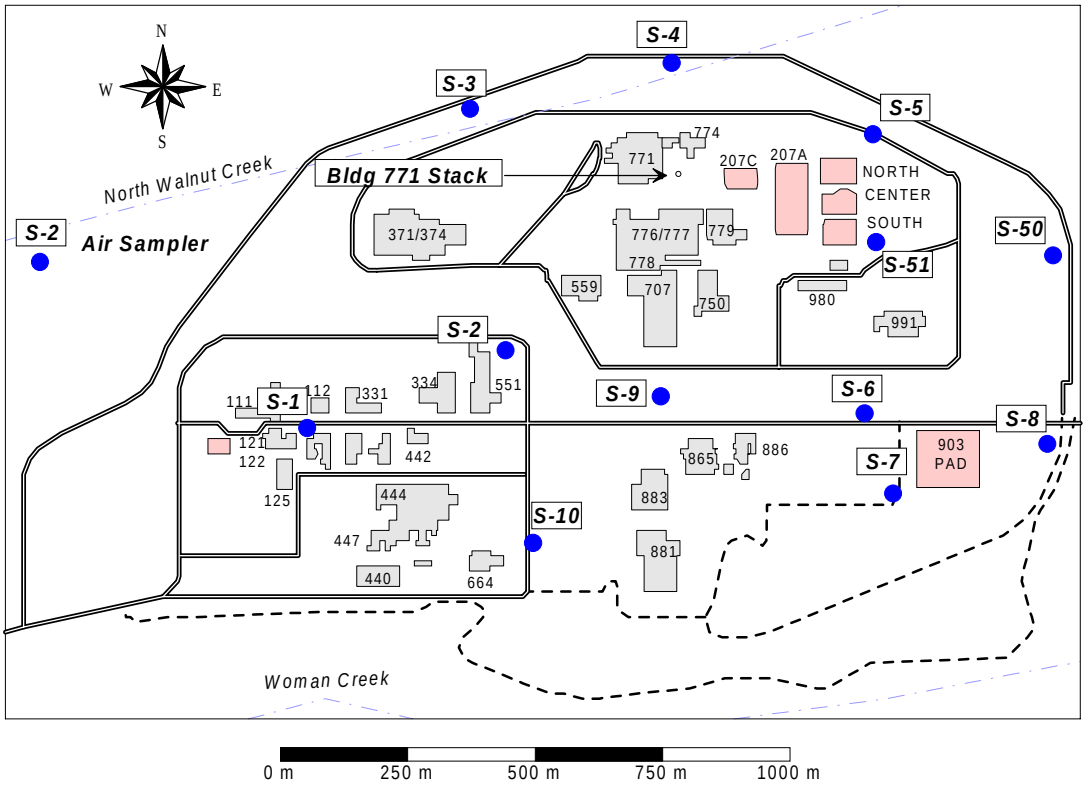

Figure 1. Main production area of the Rocky Flats Plant as it appeared in 1990. Originally, the buildings were identified with two-digit numbers. Later, a third digit was added. The production area, now sometimes called the industrial area, is surrounded by a security perimeter fence. The 44-m high Building 771 stack identified in the northern part of the facility was the primary release point from the fire. Placement of air samplers is based on air sampler locations and numbers that existed before 1973. See Chapter III and Appendix B in Rope et al. (1999) for maps of air sampler locations as they existed with respect to past features.

Through a 1989 Agreement in Principle between the DOE and the State of Colorado, DOE provided the State with funding and technical support for health-related studies. The purpose of the Historical Public Exposures Studies on Rocky Flats is to estimate exposure to nearby

residents from past toxic and radioactive releases from the plant. The Colorado Department of Public Health and Environment (CDPHE) first invited a national panel of experts to help design the health studies. Because of intense public concern about Rocky Flats contamination among Denver metropolitan area residents following a Federal Bureau of Investigation raid of Rocky Flats in June 1989, the panel decided to stress public involvement and to separate the research into two major phases conducted by two different contractors to enhance accountability and credibility.

Phase I of the study was performed by ChemRisk (a division of McLaren/Hart, Environmental Engineering). In Phase I, ChemRisk conducted an extensive investigation of past operations and releases from the RFP. The Phase I effort identified the primary materials of concern, release points and events, quantities released, transport pathways, and preliminary estimates of dose and risk to offsite individuals. The conclusions from Phase I were released in a public summary document by the Health Advisory Panel (HAP) (HAP 1993); a series of task reports by ChemRisk (ChemRisk 1994a, 1994b, 1994c. 1994d); and several articles in the journal *Health Physics*.

Radiological Assessments Corporation (*RAC*) was awarded the contract to conduct Phase II of the study, which is an in-depth investigation of the potential doses and risks to the public from historical releases from Rocky Flats. Recommendations for work to be performed in Phase II are outlined in the Phase I summary document HAP (1993).

This report documents fate and transport calculations for plutonium[1] released during the fire that occurred in Building 771 on September 11, 1957. The fire resulted in breach of the filtration system and release of plutonium from the 44-m stack. We estimated time-averaged plutonium airborne concentrations at different receptor locations within the model domain and calculated lifetime cancer incidence risks for hypothetical individuals residing in the model domain who inhaled airborne plutonium. Behavior and physical attributes of hypothetical individuals are characterized by exposure scenarios, which are discussed in detail. This report describes details of atmospheric transport modeling and uncertainty estimates, summarizes atmospheric transport calculations and risk estimates made in Phase I, and provides an overview of the source term developed for Phase II and documented in Voillequé (1999a). This report also summarizes lifetime cancer incidence risks using risk coefficients and associated uncertainty developed by Grogan et al. (1999) and reviews and discusses soil, vegetation, and air monitoring data useful for model validation.

SUMMARY OF THE GLOVE BOX FIRE IN BUILDING 771[2]

This section provides a brief summary of the events that led up to the glove box fire in Building 771, the subsequent fire, and release of plutonium. A detailed accounting of these events can be found in Voillequé (1999a), ChemRisk (1994b), and the accident investigation reports generated by Dow Chemical (Epp et al. 1957a; 1957b). The primary fire began in a glove box in Room 180 at 10:06 p.m. September 11, 1957, probably as a result of spontaneous ignition of metallic plutonium casting residues. The primary fire in Room 180 was extinguished by 10:38

[1] In this context, the word plutonium means weapons grade plutonium which consists primarily of ^{239}Pu (≈93.8%), ^{240}Pu (≈5.8%), and ^{241}Pu (≈0.36%) by weight percent. Specific activity of weapons grade plutonium is 0.072 Ci g^{-1}.

[2] Building 771 was known as Building 71 in 1957.

p.m.; however, the fire had spread to the exhaust filter plenum system, culminating in an explosion in the ventilation system at 10:39 p.m.. The exhaust filter plenum consisted of a long concrete-block walled room into which the individual exhaust systems discharged. Exhaust air was passed through a structural steel framework containing 620, 24-inch square Chemical Warfare Service (CWS) high-efficiency particulate air (HEPA) filters. Four exhaust fans connected to the filtered side of the plenum and discharged into a common exhaust duct leading to a concrete tunnel and the 44-m Building 771 stack.

The explosion in the filter plenum breached the HEPA filters, permitting the direct release of contaminants up the building stack. Brief flareups of the fire in Room 180 after the explosion were promptly put out. By 2 a.m., water had been applied to the plenum filter fire for more than 2 hours. There was no obvious burning of filters still in place or those pulled out of the holders onto the floor by fireman. Some smoldering no doubt continued (the fire was not declared "out" for several more hours), but releases from the water-soaked filters were small compared to those that occurred earlier.

Analysis, based upon post-fire experimental evidence, indicates that the flammable CWS filters used in the exhaust filter plenum system burned rapidly. It is estimated that the largest releases occurred between 10:15 p.m. and 10:45 p.m., before the primary fire was extinguished and before water was applied to the plenum filter fire. Exhaust fans were off at 10:40 p.m. when the fire burned through the power cable. After 10:45 p.m., the primary fire was out, the exhaust fans were off, and releases were much lower. The principal contributor to the releases after 10:45 p.m. was the burning filter plenum.

A summary of the chronology of the fire event follows. Refer to Voillequé (1999a) for details.

- September 11, 1957
  - 10:06 p.m. Fire started (open flame) in glove box
  - 10:09 p.m. Conveyer glove box outlet filter burned; booster system exhaust filters catch fire
  - 10:10 p.m. Fire discovered
  - 10:12 p.m. Hole burned in booster system exhaust filters
  - 10:17 p.m. Local prefilters in room exhaust system burn
  - 10:18–10:24 p.m. Heat and smoke buildup in plenum
  - 10:20 p.m. Main plenum filters begin to ignite
  - 10:25 p.m. Fans turned on high speed; CO_2 fire extinguishers ineffective
  - 10:28 p.m. Smoke detected from exhaust fan system
  - 10:29 p.m. Booster system filters consumed
  - 10:30 p.m. Plenum fire burning
  - 10:38 p.m. Fire in Room 180 is extinguished
  - 10:39 p.m. Explosion in ventilation system
  - 10:40 p.m. Power to exhaust fans shutoff; power cable in plenum is burned through
  - 10:45 p.m. Recognition of fire in main filter bank
  - 10:45 p.m. Hoses into plenum to fight fire
- September 12, 1957
  - 2:00 a.m. Filter fire declared "knocked down"
  - 11:28 a.m. Filter fire declared "out."

REVIEW OF THE PHASE I EVALUATION OF THE 1957 FIRE

In Phase I, the 1957 fire was determined to be the second largest plutonium release event at Rocky Flats. To analyze the 1957 fire, ChemRisk (ChemRisk 1994c) divided it into four main time periods:

1. From the start of the fire until the explosion
2. The explosion
3. After the explosion up to the time that the filter fire was controlled
4. From the time the filter fire was controlled until the last fire was extinguished.

Based on these four time periods a two-stage release was evaluated. The first stage was of short duration and included the start of the fire through the filter plenum explosion. This release of plutonium was characterized by coarse particles that deposited within a relatively short distance of the plant, and it did not contribute to offsite exposures. The second stage was of longer duration and was characterized by the release of fine particles during the remainder of the fire.

Environmental samples collected during and following the 1957 fire were used to estimate releases from the fire and to predict offsite exposures. Data were available from eight onsite air samplers and one offsite air sampler at a location known as Wagner School. Data were also available from portable air samplers that were deployed onsite during the fire. These data provided information about short-term average air concentrations.

The INtegrated PUFF dispersion code (INPUFF2) version 2 was used to model the atmospheric dispersion and deposition of the released plutonium. The onsite meteorological data were reported as 15- minute average wind speed and direction from 10:00 p.m. on September 11, 1957 to 10:00 a.m. on September 12, 1957. These data were reported in a Dow Chemical letter to Dr. Roy Cleare, Executive Director of the Colorado Department of Health, dated March 20, 1970. The original records for these data have never been found. Atmospheric stability was inferred for the various stages of the fire based on the wind speeds and meteorological conditions recorded at the plant site and cloud cover data recorded at Denver Stapleton International Airport and Lowry Air Force Base. Consideration was also given to the meteorological conditions known to typically prevail at Rocky Flats during the night and early morning hours.

To model the coarse particle release (stage 1), a single puff of coarse particles was assumed to have been released at 10:40 p.m. on September 11, 1957, to coincide with the explosion in the main filter plenum. A modeling interval of 1 hour was used to allow ample time for the majority of the large particles to deposit. Because particle size distribution data were not available for the coarse particles, an average deposition velocity that best fit the available vegetation sampling data and was consistent with the meteorological data was determined. ChemRisk factored uncertainty in the analytical techniques for the vegetation monitoring data, the application of the INPUFF2 model to this situation, and the limited time resolution and available data points on which to base the modeling into the analysis. An instantaneous release of 60 μCi of plutonium as coarse particles was determined. The upper and lower bounds of the 95% confidence interval about the best estimate were 1900 μCi and 1.9 μCi, respectively.

To model the fine particle release (stage 2), the INPUFF2 model used the meteorological data and atmospheric stability estimates to predict average airborne concentrations of plutonium at the locations of the air samplers and at any other locations of interest. Because the

meteorological data are in 15-minute intervals, INPUFF2 was used to predict 15-minute average air concentrations throughout the duration of the fire. First, INPUFF2 was used to develop estimates of the average releases of fine particles for two periods of the fire (between 10:40 p.m. and 2:00 a.m. and after 2:00 a.m.) that were consistent with the information obtained from the routine onsite air samplers and the offsite air sampler at the Wagner School site. After release rates that were consistent with the routine sampling information were identified, the estimated air concentrations at the locations of the portable samplers were then predicted. This provided a check to determine if there was general agreement between observations and predictions.

For the analysis, it was assumed that removal of the fine particles from the air by deposition to the ground played a minor role. A zero settling velocity and a deposition velocity of 0.1 cm s^{-1} were assumed, which is consistent with submicron-size particles. Precipitation scavenging was not taken into account for the 1957 fire because precipitation was not recorded in the region around that time. The stack height and plume rise were accounted for in the modeling.

Because the plutonium concentrations measured in many of the air samples were at or below the minimum detection limit, the Phase I release estimates were considered upper bound estimates rather than best estimates. Uncertainty in the application of the INPUFF2 model to this release situation, the sampling devices and analytical techniques for the air monitoring data, and the limited time resolution and available data points on which to base the modeling were all factored into the analysis using Monte Carlo techniques. An average release rate of fine particles of plutonium of 4 μCi s^{-1} and 0.07 μCi s^{-1} (before adjusting for uncertainties) was determined for the two separate time periods. The total estimated release of fine particles from the 13.5-hour fire event was 0.07 Ci. The upper and lower bounds of the 95% confidence interval about this total estimated release were 2.4 Ci and 0.002 Ci, respectively.

To evaluate the exposures that resulted from the releases, ChemRisk divided the study area, which extended between 2 and 10 miles radius from the center of the plant, into 24 relatively uniform sectors in all directions around the plant (see ChemRisk 1994d, Figure 3-4). However, because the variation between the predicted air concentrations at the centroid and the corners of a sector often exceeded a factor of 50, ChemRisk chose to characterize the exposures that would have occurred along the centerline of the predicted contaminant plume (that is, the highest concentrations at a particular distance from the source). In Phase I, two air concentration isopleths extending to the east (Plume A) and southeast (Plume B) from the plant were predicted for the 1957 fire. For the 1957 fire release event itself, only inhalation exposure was considered relevant. The inhalation dose estimates for $^{239/240}$Pu released during the 1957 fire are presented in Table 1 at three distances along the centerline of the two plumes, and for Denver (16 miles southeast of the plant), Lakewood (12 miles south-southeast of the plant), and Longmont (20 miles north of the plant). Pathway specific and total doses that would have resulted from the plutonium deposited during the 1957 fire were also calculated. The annual total doses from inhalation of resuspended dust and soil ingestion were four orders of magnitude lower than the inhalation doses reported in Table 1.

Preliminary cancer risk estimates were also presented in Phase I of the study. A risk coefficient of 7.3% Sv^{-1} was used based on ICRP (1990). This risk coefficient includes fatal and nonfatal cancers and severe hereditary effects. The risk estimates associated with inhalation exposure during the 1957 fire are given in Table 1.

Table 1. Phase I Dose and Risk Estimates for Airborne Releases of Plutonium During the 1957 Fire

Location – distance from plant	Inhalation dose[a] (Sv)	Risk estimate[b]
Plume A - 3 mi	1.8×10^{-5} (6.2)	1×10^{-6} (3×10^{-8} to 5×10^{-5})
Plume A - 5 mi	1.5×10^{-5} (6.2)	1×10^{-6} (3×10^{-8} to 4×10^{-5})
Plume A - 8 mi	1.2×10^{-5} (6.2)	9×10^{-7} (2×10^{-8} to 3×10^{-5})
Plume B - 3 mi	9.4×10^{-6} (6.2)	7×10^{-7} (2×10^{-8} to 3×10^{-5})
Plume B - 5 mi	5.5×10^{-6} (6.2)	4×10^{-7} (1×10^{-8} to 2×10^{-5})
Plume B - 8 mi	2.8×10^{-6} (6.2)	2×10^{-7} (5×10^{-9} to 8×10^{-6})
Denver	5.1×10^{-7} (6.2)	4×10^{-8} (1×10^{-9} to 1×10^{-6})
Lakewood	2.5×10^{-9} (6.2)	2×10^{-10} (5×10^{-12} to 7×10^{-9})
Longmont	NA[c]	NA

[a] From Appendix L, ChemRisk (1994d); geometric mean (geometric standard deviation).
[b] From Figure 5-2, ChemRisk (1994d); geometric mean (2.5% and 97.5%).
[c] NA = not applicable. Contamination plume did not reach this location

PHASE II RELEASE ESTIMATES FOR THE 1957 FIRE

Phase II release estimates (Voillequé 1999a) were substantially higher than those for Phase I. Total release quantities ranged from 11 Ci at the 5% level to 36 Ci at the 95% level. For the Phase II analysis of the 1957 fire, distributions of release quantities were reported in terms of percentiles in 5% increments for each 15-minute period of the event. The 15-minute interval was chosen to match the resolution of the meteorological data and, thereby, facilitate the calculations of environmental transport and risk. The fire event was modeled beginning at 10:00 p.m. on the evening of September 11, 1957, and ending at 2:00 a.m. the morning of September 12, 1957. The 5th, 50th, and 95th percentiles of the distribution are given in Table 2 and the reader is referred to Voillequé (1999a) for a detailed accounting of the release quantities. Most of the release was postulated to occur during the first hour of the accident as a result of the explosion in the ventilation system and filter fire. Release quantities were converted from grams of plutonium to activity (in curies) using a conversion factor of 0.072 Ci g^{-1} for weapons grade plutonium.

The sizes of plutonium particles that were released during the fire are not known. The particle size of the airborne activity was not measured. Experiments in which plutonium was oxidized and contaminated materials were burned have yielded a broad range of aerosol sizes. Release fractions that were used in the calculations considered the category of respirable particles, which are particles that may penetrate into the pulmonary region of the lung. The maximum aerodynamic diameter of particles that can enter the deep lung is about 10 µm. Because the particle size distribution is not known, we considered a range of aerodynamic equivalent diameters (AED) between 1 and 10 µm in the transport and risk calculations.

As noted earlier, the release estimates made for Phase II are considerably larger than for those of Phase I, which relied on environmental monitoring data in vegetation to estimate source terms. The Phase II source term was based on an examination of the plutonium that would have been trapped on the HEPA filters and deposited in the exhaust ventilation system.

Table 2. Summary of 1957 Fire Plutonium Source Term Reported in Voillequé (1999a)

Date	Time	Release quantity (Ci)		
		5%	50%	95%
September 11, 1957	10:00–10:15 p.m.	3.4×10^{-2}	6.0×10^{-2}	9.9×10^{-2}
	10:15–10:30 p.m.	$5.7 \times 10^{+0}$	$1.1 \times 10^{+1}$	$1.7 \times 10^{+1}$
	10:30–10:45 p.m.	$4.1 \times 10^{+0}$	$8.8 \times 10^{+0}$	$1.6 \times 10^{+1}$
	10:45–11:00 p.m.	1.4×10^{-1}	2.8×10^{-1}	5.0×10^{-1}
	11:00–11:15 p.m.	1.4×10^{-1}	2.7×10^{-1}	4.9×10^{-1}
	11:15–11:30 p.m.	6.7×10^{-2}	1.4×10^{-1}	2.6×10^{-1}
	11:30–11:45 p.m.	6.7×10^{-2}	1.3×10^{-1}	2.5×10^{-1}
	11:45–00:00 a.m.	6.7×10^{-2}	1.3×10^{-1}	2.5×10^{-1}
September 12, 1957	00:00–00:15 a.m.	3.0×10^{-2}	6.6×10^{-2}	1.4×10^{-1}
	00:15–00:30 a.m.	3.8×10^{-2}	7.5×10^{-2}	1.4×10^{-1}
	00:30–00:45 a.m.	2.4×10^{-2}	4.8×10^{-2}	8.3×10^{-2}
	00:45–01:00 a.m.	1.7×10^{-2}	3.4×10^{-2}	5.9×10^{-2}
	01:00–01:15 a.m.	1.0×10^{-2}	2.1×10^{-2}	3.8×10^{-2}
	01:15–01:30 a.m.	6.9×10^{-3}	1.5×10^{-2}	2.7×10^{-2}
	01:30–01:45 a.m.	4.3×10^{-3}	9.5×10^{-3}	1.9×10^{-2}
	01:45–02:00 a.m.	1.5×10^{-3}	4.4×10^{-3}	1.1×10^{-2}
TOTAL		$1.1 \times 10^{+1}$	$2.1 \times 10^{+1}$	$3.6 \times 10^{+1}$

ENVIRONMENTAL MONITORING OF PLUTONIUM

Historical environmental monitoring data relevant to assessing contaminant emissions from the Rocky Flats Plant are evaluated in Rope et al. (1999). This section briefly reviews data that are pertinent to the release of plutonium from the 1957 fire event, emphasizing data that may potentially be used for model validation. This limits our discussion to measurements of plutonium in ambient air and deposition on vegetation. Most soil measurement data were taken after 1970 and, evidence of deposition from the 1957 fire is obscured by subsequent releases, in particular deposition from 903 Area releases. In general, we found air monitoring data to be of little use for validating model predictions because

- Air monitoring practices in 1957 used the 4-hour delay counting technique. Short-lived alpha activity from natural sources (radon progeny) and poor detection limits tended to obscure activity originating from the RFP.
- The 4-hour delay counting technique has shown poor correlation to the preferable 1-week count of long-lived alpha activity.

Ambient Air Monitoring

A detailed review and analysis of plutonium monitoring in air around Rocky Flats is documented in Rope et al. (1999). Air monitoring was performed by the site contractor and several other independent agencies. Before 1970, samplers were only analyzed for total long-lived alpha activity (TLLa). The RFP contractor began onsite ambient air monitoring at a single station in 1952. By early 1953, 10 onsite stations (Figure 1) had been established; in 1969, two

additional stations were added. During the 1957 fire, eight onsite samplers (S-1 through S-8) and one offsite sampler located at the Wagner school (≈5.3 km southeast of the plant) were operating (ChemRisk 1994c). In addition to these routine air samplers, portable samplers were deployed at 13 onsite locations during the fire. Sampling results were reported as the percent of the maximum permissible level (MPL) of 9 disintegrations per minute (dpm). The routine air monitoring equipment were low volume air samplers that operated at a flow rate of 0.0566 m^3 min^{-1} (2 ft^3 min^{-1}). Portable high-volume samplers were maintained for events like the fire and operated at a flow rate of 0.0339 to 1.13 m^3 min^{-1} (12 to 40 ft^3 min^{-1}). Onsite sampler filters were typically changed daily at 8:15 a.m. However, following the fire on the evening of September 11th, onsite sampler filters were changed at 3:30 a.m. and again at 3:30 p.m. of the following day (September 12th). Filters of offsite samplers were reported to be changed biweekly, but it was unclear whether this meant twice a week or once every two weeks (ChemRisk 1994c). The filter on the Wagner school sampler that would have captured releases from the fire was installed on September 10 at 11:15 a.m. This filter was changed at 2:15 a.m. on September 12th and again sometime on September 13th; however, the exact time of replacement was not reported.

A review of the air monitoring data revealed inconsistencies between activity detected and the timing of the release. For example, the highest air concentrations from the portable sampling network appear to have occurred the morning of September 12th at around 6:00 a.m., well after the fire had been put out and the major release had occurred. The onsite samplers (S-1 through S-8) reported concentrations of 0% of the MPL for the period 8:15 a.m. September 11 to 3:30 a.m. September 12. After that, an average of the onsite samplers operating from 3:30 a.m. to 3:30 p.m. September 12 reported a concentration of 0.28% of the MPL (ChemRisk 1994c). It is postulated that during the main release from fire (10:15 to 10:45 p.m.), the plume was lofted over onsite locations resulting in low ground-level concentrations. It is not clear why measurable activity was detected in the samplers for the later measurement period. Similar discrepancies were noted for the Wagner School air sampler. A concentration of 0% MPL was reported for the time period September 10, 11:15 a.m. to September 12, 2:15 a.m. The second filter that was removed sometime on September 13 reported a concentration of 0.56% MPL.

Perhaps of greater importance is the assessment of the overall quality of the air monitoring data reported in Rope et al. (1999). Rope states that in the 1950s (particularly 1955–1960), 4-hour gross alpha counts were made. The count was made 4 hours after collection and included large contributions from natural alpha emitting radionuclides like radon decay products. Rope concludes that the 4-hour count results are of no value in assessing the concentrations of long-lived alpha emitters released from Rocky Flats.

Air monitoring during the 1957 fire as reported by ChemRisk (1994c) were carefully reviewed for applicability towards model validation of predicted air concentrations made in this report. We also considered the data quality issues discussed in Rope et al. (1999). Based on the data quality issues discussed in Rope et al. and discrepancies between the timing of the event and the measurements, we conclude that the air monitoring data are of little use in validating model predictions.

Vegetation Monitoring

Vegetation is the only media for which a significant number of plutonium-specific measurements were made before 1970. This media may, therefore, be important for validating

source terms and environmental transport modeling. Monitoring of vegetation began before the site was operating as part of a preoperational background study. Initial monitoring began in 1952, but it was ended in 1953 because of technical problems. Vegetation monitoring resumed after the 1957 fire. In response to the fire, samples were collected at all the locations established in the 1952–1953 vegetation monitoring program (Figure 2). The samples collected in conjunction with the fire included 222 vegetation samples in 1957 and 132 vegetation samples in 1958. Over 80 of the 1957 samples were analyzed specifically for plutonium.

The data shown in Figure 2 represent only the gross alpha measurements made during one of the sampling surveys performed in 1957 following the fire. The Site Survey-Monthly Progress Reports, in which these data were reported, do not specify the types of plants that were collected or the portions of the plants that were analyzed. It appears that the collection and analytical techniques were the same as those used in 1952 and 1953 (Hammond 1957, 1958). The Environmental Survey Reports present the average and maximum values for all samples collected within a given distance range from the plant.

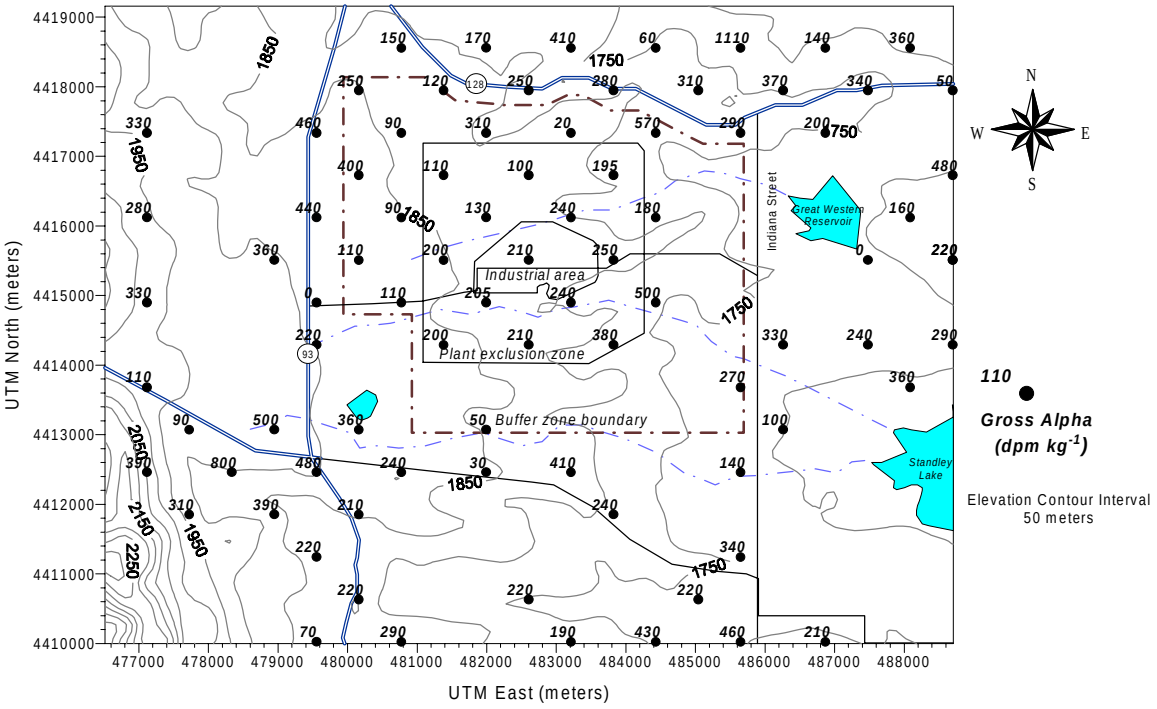

Figure 2. Gross alpha activity in vegetation measured after the fire from September 13 to December 17, 1957. Measurements are not corrected for background activity levels or activity deposited before the fire.

Vegetation monitoring data are potentially useful for model validation. However, predicting concentrations on vegetation requires an additional level of modeling to compute deposition and retention over time on vegetative surfaces. With this additional level of modeling comes additional uncertainty. Only crude, generalized vegetation models can be used for comparison because the types of plants sampled and other factors (such as vegetative cover and soil conditions that affect redistribution of deposited activity between the soil and plant surface) were not specified. Furthermore, atmospheric transport models lack the spatial temporal resolution to

make a point-by-point comparison of vegetation concentrations, given that vegetation concentration are highly variable and depend on local conditions that are unknown. Therefore, we expect comparisons to consist only of general temporal trends of activity on vegetation. Prediction uncertainty is expected to be quite large (a factor of 10 to 100).

ENVIRONMENTAL TRANSPORT MODELING

Offsite exposure to plutonium from releases resulting from the September 1957 glove box fire and subsequent breach of the HEPA filtration system in Building 771 were investigated in Phase I and are summarized in a previous section of this report. Airborne releases were considered to be the principal transport pathway and inhalation the major pathway of exposure.

Atmospheric releases of plutonium as a result of the glove box fire primarily occurred from the 44 m Building 771 stack. This section describes our approach to estimating atmospheric dispersion of plutonium released from this event and the uncertainty associated with concentration estimates in the model domain. Our approach to this calculation involves first estimating the plume trajectory based on the available meteorological data. Next, the stochastic source term developed by Voillequé (1999a) is coupled with the dispersion model to generate concentration isopleths in the model domain, incorporating uncertainties in the dispersion process. Distributions of airborne concentrations are then used with exposure scenarios and plutonium inhalation risk coefficients to calculate incremental lifetime cancer incidence risk for hypothetical receptors in the model domain.

Atmospheric Model Selection

Five atmospheric transport models considered for use in this study were evaluated in Rood (1999a): (1) the Terrain-Responsive Atmospheric Code (TRAC) (Hodgin 1991), (2) the Industrial Source Complex Short Term Version 2 (ISC) (EPA 1992), (3) Regional Atmospheric Transport Code for Hanford Emission Tracking (RATCHET) (Ramsdell et al. 1994), (4) TRIAD (Hicks et al. 1989), (5) and INPUFF2 (Petersen and Lavdas 1986). The model comparison study determined what models, if any, performed best in the Rocky Flats environs for a given set of modeling objectives.

Model evaluations were based on how well predictions compared with measured tracer concentrations taken during the Winter Validation Tracer Study (WVTS) (Brown 1991) conducted in February 1991 at the RFP. The WVTS consisted of 12 separate tests: 6 tests were conducted during nighttime hours, 4 during daytime hours, and 2 during day-night transition hours. For each test, an inert tracer (sulfur hexafluoride) was released at the RFP at a constant rate for 11 hours from a 10-m high stack located on the southern boundary of the RFP industrial area. Two sampling arcs, 8 and 16 km from the release point, measured tracer concentrations every hour for the last 9 hours of each test period. Seventy-two samplers were located on the 8-km arc, and 68 samplers were located on the 16-km arc. Predicted concentrations were then compared to the observed tracer concentrations at each of the samplers.

We acknowledge that the release conditions of the WVTS are substantially different from the glove box fire release conditions. Most notably, the release height and release temperature. Of particular concern is the possibility that glove box fire releases were entrained in an upper-level air mass that showed little resemblance to the surface-level air mass where the meteorological

measurements were made. If this was the case, then we have little hope in reconstructing the plume trajectory because upper-air meteorological data do not exist. Despite these shortcomings, the WVTS is the most complete site-specific data set available with which to evaluate atmospheric transport models. We can expect that an elevated release will increase the uncertainty in a model prediction, but we have assumed that the relative performance among models was adequately characterized by the comparison with the WVTS data.

Modeling objectives for the comparison study were based on the premise that identifying hourly locations of individual receptors was unlikely. Instead, it was more likely to identify receptors (hypothetical or real) who were present at a fixed location for the duration of a release event. The minimum time scale of historical release events at RFP ranged from 6–10 hours to several days. Release events modeled for the WVTS were 9 hours in duration. If we assume the receptor is fixed for a time period of at least 9 hours, then the time-averaged concentration (9-hour average) is an appropriate modeling objective rather that comparing hourly average concentrations. Therefore, models were evaluated based on their performance in predicting time-averaged concentrations at fixed sampler locations in the model domain (9-hour average concentration at each sampler paired with the corresponding predicted value). We also considered the arc-integrated concentration. The arc-integrated concentration was the 9-hour average ground-level concentration integrated across the 8- and 16-km sampling arc. The latter performance objective provides a measure of the vertical dispersion component of the models and the ground-level tracer mass, 8 and 16 km from the release point. Data sets for the time-averaged concentration were limited to only those points where the predicted (C_p) and observed (C_o) concentration pair were greater than the time-averaged minimum detectable concentration.

Fifty percent of the time-averaged model predictions were within a factor of 4 of the observations. Predicted-to-observed ratios (C_p/C_o) ranged from 0.001 to 100 and tended to be higher at the 16-km arc than the 8-km arc. Geometric mean (GM) C_p/C_o ratios ranged from 0.64 (TRAC) to 1.5 (ISC), and geometric standard deviations (GSDs) ranged 4.4 (RATCHET) to 6.5 (ISC). The RATCHET model had the highest correlation coefficient for the 8-km (0.67) and 16-km (0.58) sampling arc, followed by TRIAD and INPUFF2 (Figure 3). Qualitatively, the predictions made by the RATCHET model appear to best match the observations. The slope of the regression line was closest to that of the perfect correlation line (solid line in Figure 3).

Arc-integrated results showed INPUFF2 and TRIAD had the highest correlation coefficients, but correlation coefficients were not significantly different (at the 95% level) from the other models. The ISC model tended to overpredict arc-integrated concentration, and the TRAC model showed the greatest variability.

The results reported in Rood (1999a) indicated no one model clearly outperformed the others. However, the RATCHET, TRIAD, and INPUFF2 models generally had lower variability (indicated by lower GSDs of C_p/C_o ratios) and higher correlation coefficients compared to those of ISC and TRAC models. It is desirable in a study such as this to choose a model that has the least amount of variability when comparing model predictions to observations. In addition, the model selected should have a level of complexity that is consistent with available data. The TRAC model is the most complex in terms of its treatment of the atmospheric dispersion process in complex terrain, but the study showed model performance was no better than the other models. In addition, the availability of meteorological data needed to fully use the capabilities of the TRAC model are lacking. The straight-line Gaussian plume model, ISC, tended to overpredict concentrations and was also limited to only one meteorological recording station in the model

domain. Available meteorological data for this study period included two meteorological recording stations: one at the RFP and the other at Denver Stapleton International Airport. Therefore, it is desirable to use a model that may include multiple meteorological recording stations in the model domain. Using multiple meteorological recording stations allows for a spatially varying wind field in the model domain.

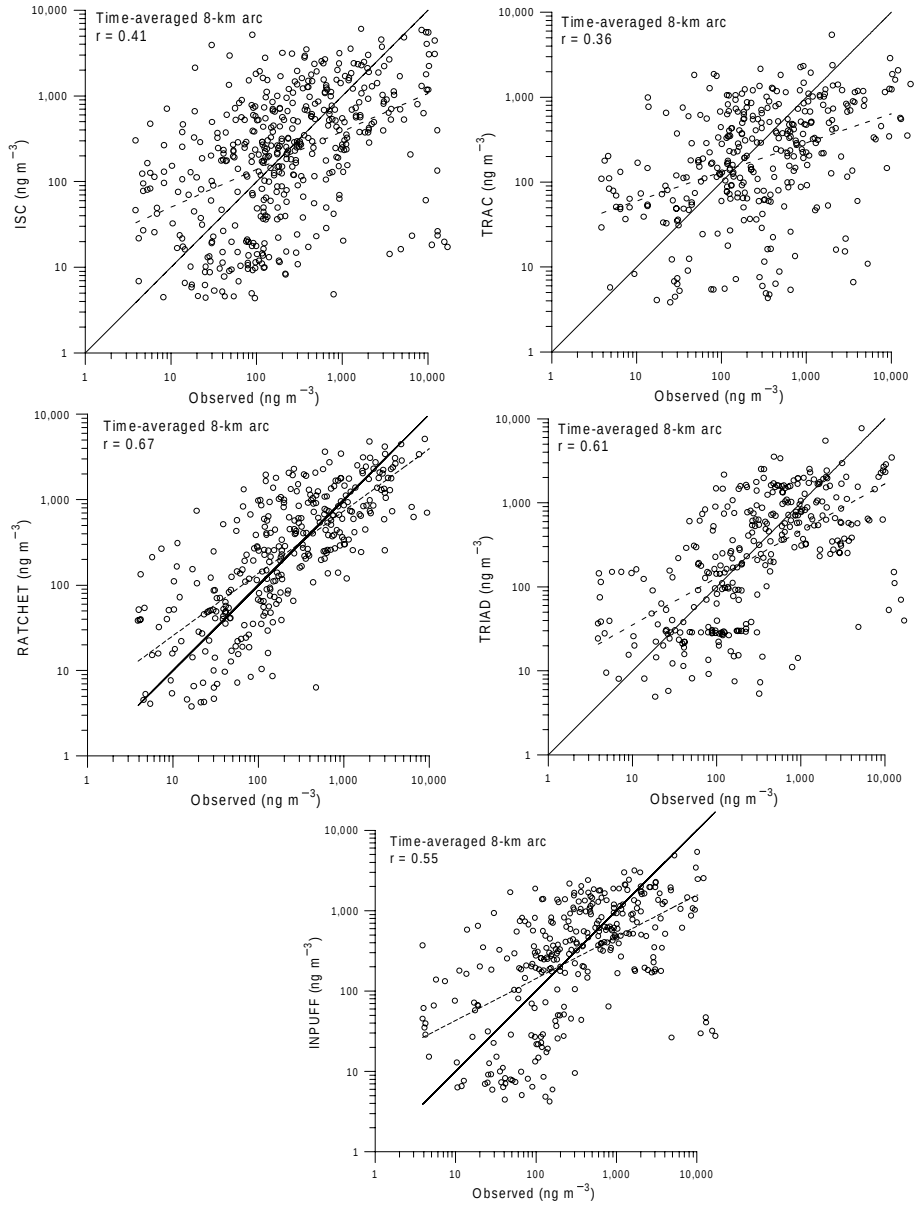

Figure 3. Nine-hour average observed concentrations as a function of predicted values for the five models compared using the WVTS data set. Correlation coefficients were for the log-transformed data. The solid line represents perfect correlation between predicted and observed values. The dashed line represents the log-transformed regression fit.

The RATCHET, INPUFF2, and TRIAD models performed comparably and were considered viable candidates for atmospheric dispersion estimates. Of these models, RATCHET and TRIAD

were chosen for more detailed evaluation because both these models allow for spatially varying wind fields.

There are advantages and disadvantages to using either model in dispersion calculations. The TRIAD model is capable of incorporating meteorological data on a user-defined time scale, while RATCHET uses a fixed, 1-hour increment. Meteorological data from the RFP during the fire were reported every 15 minutes; therefore, the TRIAD model would appear better suited for the calculation because it could incorporate the resolution of the meteorological data. However, Denver Stapleton International Airport data were recorded every hour, so this advantage is lost in terms of predicting wind vectors at that distance. In addition, model comparisons using the WVTS data set showed that RATCHET performed as well, if not better than TRIAD, using 1-hour average meteorological conditions. RATCHET also has several other features that make it desirable, including

- Spatial varying surface roughness lengths and mixing heights
- Algorithms to compute plume depletion and deposition for fine particles are included (deposition must be computed outside the TRIAD codes)
- Random sampling routines that facilitate Monte Carlo calculations.

We chose the RATCHET model to perform the calculations based on its performance in the WVTS model comparison (Rood 1999a) and the features of the code stated previously. Features of the RATCHET model are summarized in Table 3.

Model Domain and Receptor Grid

The model domain (Figure 4) encompasses a 2200 km^2 area (50 km north-south × 44 km east-west). The domain extends 28 km south, 12 km west, 22 km north, and 32 km east from the RFP. Most of the Denver metropolitan area and the city of Boulder are included in the domain. The domain was limited in its western extent because few receptors are present there and most of the contaminant plumes traveled east and southeast of the plant.

RATCHET uses two modeling grids. Hourly meteorological records are used to estimate wind speed and direction, stability, and precipitation on the environmental grid in addition to surface roughness features. The concentration grid has spacing one-half that of the environmental grid. Ground-level concentrations and deposition are output at each of these grid nodes. The environmental grid was set at 23 nodes east-west and 26 nodes north-south, with a grid spacing of 2000 m. The concentration grid has 45 nodes east-west and 51 nodes north-south, with a spacing of 1000 m. The southwest corner of the model domain has the universal transverse mercator (UTM) coordinates 470850 E and 4387050 N. Release points are defined by distances (in kilometers) from a reference node. The reference node for the environmental grid was (7,15), and the reference node for the concentration gird was (13,29); they both have the UTM coordinates of 482850 E and 4415050 N.

Figure 4 was generated using U.S. Geological Survey (USGS) 7.5-minute digital elevation models. Topographic contours were based on an elevation grid spacing of 100 m. Major roadways and water features were digitized from USGS 1:100,000 digital line graphs.

Table 3. Features of the RATCHET Model

Feature	Representation in RATCHET
Domain area[a]	2200 km^2
Node spacing[a]	2000 m
Source term	Hourly release rates
Meteorological data	Hourly
Surface roughness	Spatially varying
Wind fields	$1/r^2$ interpolation (r = the radial distance from the observation)
Topographical effects	None explicit[b]
Wind profile	Diabatic
Stability	Spatially varying based on wind, cloud cover, and time of day
Precipitation	Spatially varying, three precipitation regimes with different precipitation rate distributions
Mixing layer	Spatially varying, based on calculated values for each meteorological station
Plume rise	Briggs' equation (Briggs 1969, 1975, 1984)
Diffusion coefficients	Based on travel time and turbulence levels
Dry deposition	Calculated using resistance model
Wet deposition	Reversible scavenging of gases, irreversible washout of particles
Model time step	15 minute maximum, 15 second minimum
Output frequency	Hourly
Uncertainty	Options available for Monte Carlo simulation within the code

[a] Modified from the original RATCHET specification for use at Rocky Flats.
[b] Terrain differences are not a model input. However, topographical influence on the wind field may be accounted for by incorporating multiple meteorological stations in the model domain.
[c] RATCHET was modified to accommodate hourly output. The original code output time-integrated concentrations daily.

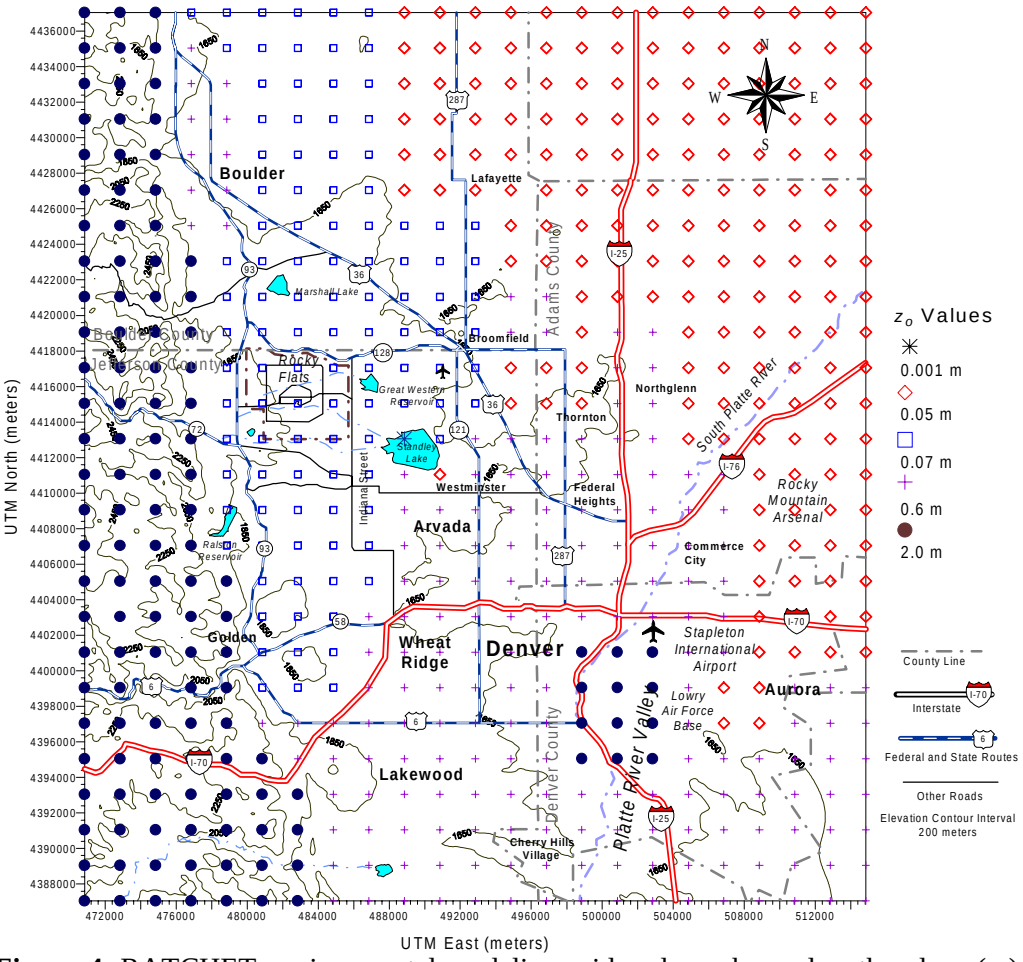

Figure 4. RATCHET environmental modeling grid and roughness length values (z_o). Symbols represent grid nodes and the z_o value assigned to the node. Roughness lengths were based on topographic contours, and urban density in the 1960s and 1970s.

Meteorology

Rocky Flats meteorological data for its operational period (1953–1989) are sporadic, incomplete, and of questionable integrity. Requests for meteorological data from the RFP were initially made by ChemRisk during Phase I of the project. ChemRisk was able to locate two letters from Dow Chemical to Dr. Roy Cleare, Executive Director of the Colorado Department of Health, dated March 20, 1970. These letters contained wind speed and direction for varying time increments during the 1957 and 1969 fire incidents. Computer diskettes containing wind speed, wind direction, and precipitation measurements from October 1968 to May 1969 were also obtained. These data were hourly observations taken approximately 15 minutes before the top of the hour and do not represent hourly average readings. Although these data appeared to be climatologically reasonable, no records of instrument calibration or audits of the information were found. Parameter resolution was very coarse (for example, wind direction resolution was 45 degrees). Original records, including the strip recording charts, were not located for the period from 1952–1983.

An extensive data search was initiated in 1994 by *RAC* researchers to locate missing data and interview personnel who were involved with measurements at the site. No new data were recovered, but several personnel reported problems with the recording instrumentation at the RFP (for example, the measured wind direction was off by 180 degrees). In 1983, a 61-m tower was constructed near the southern boundary of the RFP industrial area. Meteorological instrumentation was installed at 10-, 25-, and 61-m heights. These instruments were coupled with digital data recorders that allowed data to be taken continuously and processed and stored on a 15-minute interval. Operation of the tower began in 1984 and data recording adhered to strict quality assurance standards. Data from 1989–1993 were used in conjunction with data from Denver Stapleton International Airport to estimate annual average dispersion factors (Rood 1999b).

In 1994, the RFP hired a subcontractor to compile, screen, validate, and analyze historical climatological data (DOE 1995). A draft report issued in February 1995 contained monthly and annual summaries of wind speeds, wind directions, precipitation, temperature, and other parameters for the years 1953–1993. While these data are of interest and may be important for some aspects of modeling, they lacked the resolution required for detailed atmospheric transport modeling.

Except for the March 20, 1970, letter from Dow Chemical to Dr. Roy Cleare, meteorological records from Rocky Flats did not cover the 1957 glove box fire. Other options were considered such, as using typical meteorological conditions for the month of September for years where reliable meteorological data were recorded (1984–present). However, while this technique is suitable for long-term dispersion estimates (as was done for the routine release assessment [Rood 1999b]), it was not viable for estimating conditions during a relatively brief event like the 1957 glove box fire because daily conditions are highly variable. Therefore, we had little recourse but to use the meteorological data provided in the Cleare letter despite its questionable nature. Data from Denver Stapleton International Airport covering the period of interest (September 11 and 12, 1957) were also obtained and used in conjunction with the RFP data for air dispersion calculations. These data were instantaneous measurements, not hourly averages as was typical of all airport data before the Automatic Surface Observation Site system was installed at most major airports. The Denver Stapleton International Airport meteorological station was located 24 km east and 14 km south from the center of the model domain (RFP). These data included measurements of wind speed, wind direction, cloud cover, and precipitation. It is known that meteorological conditions in the Denver metropolitan area can differ significantly from those at Rocky Flats (DOE 1980). Therefore, it is unreasonable to use meteorological data from Denver alone for simulations involving releases from Rocky Flats. In these simulations, initial plume trajectories are primarily influenced by the wind direction at Rocky Flats. Only after plume elements are transported to the Denver metropolitan area are trajectories and dispersion influenced by meteorological conditions in Denver and represented by Denver Stapleton International Airport data.

Data Processing

Meteorological data from the Cleare letter were obtained from the Phase I Task 6 report (ChemRisk 1994c, Appendix F). No mention of the instrument height was provided; therefore, we assumed measurements were made at the 10-m level, which is the typical meteorological

measurement height. Wind speed was extrapolated to the release height within the RATCHET code using a diabatic wind speed profile. Measurements were reported on 15-minute intervals and included wind speed, wind direction, and atmospheric stability. Mixing layer depths were not provided. Mixing-layer depths are calculated hourly within RATCHET at each active meteorological recording station using a methodology described by Zilitinkevich (1972). The RATCHET code also requires default mixing layer depths for each month, stability class, and hour of day. These data were compiled from processed meteorological data taken at the RFP 61-m tower from 1989 to 1993. The calculated or default value is selected based on the relative magnitude of the calculated and default values, the stability, season, and time of day. The larger of the two is selected for the meteorological recording station for the given hour. A multiple linear regression technique is then used to provide a smooth spatial variation in mixing-layer depth across the model domain.

Hourly stability classes were calculated separately for the RFP and Denver Stapleton International Airport meteorological recording stations using the general classification scheme discussed in Pasquill (1961), Gifford (1961), and Turner (1964). This typing scheme employs seven stability categories ranging from A (extremely unstable) to G (extremely stable) and requires estimates of cloud cover and ceiling height. Cloud cover and ceiling height data for both stations were assumed to be the same and were obtained from the Denver Stapleton International Airport data.

Hourly average wind speed and direction also were calculated from the RFP meteorological data using the protocol described in U.S. Environmental Protection Agency (EPA) (1987). An arithmetic average of the wind direction was computed first, and it was then segregated into 1 of 36, 10-degree increments as required by RATCHET. The average wind speed for the hour was computed by taking the average of the four, 15-minute data segments. No precipitation was recorded in the model domain for the duration of release and subsequent transport and dispersion of material out of the model domain.

One modification was performed to the data provided in the Cleare letter. For the first hour of the simulation (10:00 p.m.–11:00 p.m.), the wind speed was changed to the average of the last two 15-minute segments (10:30 and 10:45). We used meteorological conditions that reflected the last 30 minutes of the hour because that is when the bulk of the release is postulated to have occurred. This resulted in a change in the mean hourly average wind direction from 106 degrees to 52 degrees. The mean wind speed remained about 1 m s^{-1} regardless of the averaging period. Meteorological data used in the simulation are summarized in Appendix A.

Atmospheric Transport Model Parameters

This section describes the input parameters we selected for the RATCHET model simulations involving transport and dispersion of plutonium released from the glove box fire in 1957. These parameters include surface roughness length, topography, dry and wet deposition, diffusion coefficients, release parameters (location and height of release), and model control parameters (number of puffs per hour and computational options).

Surface Roughness Length

Roughness elements (such as trees and buildings) and small-scale topographic features (such as rolling hills) have a frictional effect on the wind speed nearest the surface. The height and spacing of these elements determine the frictional effects on the wind. These effects are directly related to transport and diffusion and affect atmospheric stability, wind profiles, diffusion coefficients, and the mixing-layer depth. The surface roughness length parameter is used to describe these roughness elements and is a characteristic length associated with surface roughness elements (Table 4). In RATCHET, estimates of the surface roughness length are defined for each node on the environmental grid (Figure 4). In our simulations, we selected a value of 0.6 m to represent residential and urban environs. Farmland, which is predominant in the northeast part of the model domain, was assigned a value of 0.05 m. Range and open land consisting of rolling grass hills were assigned a value of 0.07 m. Nodes that encompass the range and farmland designation were selected based on the topographic contours and land use maps from the 1960s and 1970s (ChemRisk 1994e). The foothills and downtown Denver were assigned a value of 2.0 m, and the large open water body (Standley Lake) was assigned a value of 0.001 m.

Table 4. Typical Surface Roughness Lengths for Different Land Use, Vegetation, and Topographic Characteristics[a]

Land use, vegetation, and topographic characteristics	Surface roughness length, z_o (m)
Level grass plain	0.007–0.02
Farmland	0.02–0.1
Uncut grass, airport runways	0.02
Many trees/hedges, a few buildings	0.1–0.5
Average, North America	0.15
Average, U.S. Plains	0.5
Dense forest	0.3–0.6
Small towns/cities	0.6–2.5
Very hilly/mountainous regions	1.5+

[a] Source: Stull (1988), Figure 9.6.

Topography

The RATCHET model does not explicitly address terrain differences within the model domain. Instead, topography and topographic effects on transport and diffusion are reflected in the surface roughness lengths and observed wind velocity data that are affected by topographical features. Topography in the model domain (Figure 4) can be characterized by three major features: the north-south trending Colorado Front Range foothills in the western part of the model domain, the southwest to northeast trending Platte River Valley located in the southeast part of the model domain, and rolling hills and flat farmland that are predominant in the central and northeastern part of the model domain. The topography generally slopes east from Rocky Flats, dropping 200 m in elevation to the Platte River Valley. The surface roughness lengths

reflect these features as stated in the previous section. Observed meteorological data are lacking in most of the model domain and are woefully inadequate to characterize wind fields in the foothills region. However, meteorological observations at Denver Stapleton International Airport do capture the air movement within the Platte River Valley, which is noticeably different than that at the RFP (DOE 1980). Therefore, to a limited extent, topography is accounted for the model simulation. The use of a complex terrain model would also suffer from the lack of meteorological data, especially in the foothills region. This region is of lesser importance because few receptors were present in the foothills when the plant was operating.

Dry Deposition and Gravitational Settling

The rate of deposition of small particles on surfaces in the absence of precipitation is proportional to the concentration of material near the surface. The proportionality constant between the concentration in air and the flux to the ground surface is the dry deposition velocity. The current generation of applied models estimates deposition using an analogy with electrical systems as described by Seinfeld (1986). The deposition is assumed to be controlled by a network of resistances, and the deposition velocity is the inverse of the total resistance. Resistances are associated with atmospheric conditions; physical characteristics of the material; and the physical, chemical, and biological properties of the surface. Originally, RATCHET was designed to calculate deposition from small particles (≈ 1μm) and reactive gases. For these small particles, gravitational settling is negligible and therefore, RATCHET excluded this process in the deposition velocity. Because particle size for the 1957 fire releases may have been substantially larger than submicron particles, gravitational settling needed to be included. We replaced the deposition model in RATCHET with one that includes gravitational settling. This model is based on the work of Slinn and Slinn (1980) and Pleim et al. (1984). We have used the formulation as presented in the ISC3 code (EPA 1992) in out formulation. The approach is similar to that used in RATCHET, but includes the effects of gravitational settling.

The total resistance including gravitational settling effects in is made up of three components: aerodynamic resistance, surface-layer resistance, and transfer resistance. Thus, the dry deposition velocity (v_d, m s^{-1}) is calculated using

$$v_d = \frac{1}{r_a + r_d + r_a\, r_d\, v_g} + v_g \qquad (1)$$

where

r_a = aerodynamic layer resistance (s m^{-1})
r_d = deposition layer resistance (s m^{-1})
v_g = gravitational settling velocity (m s^{-1}).

Note that for large settling velocities, the deposition velocity approaches the settling velocity, whereas, for small settling velocities, v_d tends to be dominated by the r_a and r_d resistance terms.

The lowest few meters of the atmosphere can be divided into two layer: a fully turbulent region where vertical fluxes are nearly constant, and the thin quasi-laminar sublayer. The resistance to transport through the turbulent, constant flux layer is aerodynamic resistance. It is usually assumed that the eddy diffusivity for mass transfer within this layer is similar to that for

heat. The atmospheric resistance formulation is based on Byun and Dennis (1995) and for stable conditions is given by

$$ra = \frac{1}{k\,u_*}\left(\ln\frac{z_d}{z_o} + 4.7\frac{z}{L}\right) \tag{2}$$

and for unstable conditions by

$$r_a = \frac{1}{k\,u_*}\left(\ln\frac{\left(\sqrt{1+16\left(z/|L|\right)}-1\right)\left(\sqrt{1+16\left(z_o/|L|\right)}+1\right)}{\left(\sqrt{1+16\left(z/|L|\right)}+1\right)\left(\sqrt{1+16\left(z_o/|L|\right)}-1\right)}\right) \tag{3}$$

where

u_* = friction velocity (m s^{-1})
k = von Karman constant (0.4)
L = Monin-Obukhov length (m)
z = height above ground surface (m)
z_o = surface roughness height (m)
z_d = deposition reference height (m)

The deposition reference height (z_d) is assigned a fixed value of 1.0 m in ISC3. We found that under conditions where z_o is greater than 1.0 m, negative deposition velocities were calculated. To alleviate this problem, we set z_d to 1.0 m + z_o. This change has little impact for small z_o values and results in positive deposition velocities for z_o's greater than 1.0 m as is found in the foothills and downtown Denver.

The deposition layer resistance was taken from the approach proposed by Pleim et al. (1984) and modified to include Slinn's (1982) estimate for the inertial impaction term.

$$r_d = \frac{1}{\left(Sc^{-2/3} + 10^{-3/St}\right)u_*} \tag{4}$$

where

Sc = Schmidt number ($Sc = \upsilon/D_B$)
υ = viscosity of air (0.15 $cm^2\ s^{-1}$)
D_B = Brownian diffusivity of the particle in air ($cm^2\ s^{-1}$)
St = Stokes number ($St = v_g/g\ [u_*^2/\upsilon]$, dimensionless)
g = acceleration due to gravity (981 cm s^{-2})

Stokes law gives the gravitational settling velocity (cm s^{-1}) for particles less than 30 µm as

$$v_g = \frac{C_c\,d^2\,g\rho}{18\mu_{air}} \tag{5}$$

where

C_c = the Cunningham slip correction factor (dimensionless)
d = physical particle diameter (cm)
μ_{air} = dynamic viscosity of air (1.78×10^{-4} g s^{-1} cm^{-2})

ρ = particle density (2.65 g cm^{-3} for soil).

For particle sizes less than several microns, the Cunningham Slip correction factor is approximately 1.0. The Brownian diffusivity (cm s^{-1}) of the particle in air is given by

$$D_B = 8.09 \times 10^{-10} \frac{C_c \, T_a}{d} \quad (6)$$

where
T_a = air temperature (°K)

Figure 5 presents gravitational settling velocity as a function of the physical diameter of the particle. Routine effluent containing plutonium was reported to pass through HEPA filtration, resulting in the release of particles less than 1 µm in diameter. Median particle size for routine effluent has been estimated to be 0.3 µm (Voillequé 1999b). Whicker and Schultz (1982) reports that gravitational settling velocities for particles less than 1 µm are insignificant compared to the other components of deposition. However, the HEPA filtration system was breached during the 1957 fire incident, allowing unfiltered plutonium to be released. Previous investigators cited in the Phase I report (Kathren 1974; Martell 1975) report the airborne fraction of burning plutonium to range from submicron (0.03 µm AED) up to 29 µm AED. Release estimates discussed in an earlier section and detailed in Voillequé (1999a) represent only the respirable fraction, which included particles up to 10 µm AED. Details of the effluent particle size distribution are lacking (other than the particles were <10 µm AED).

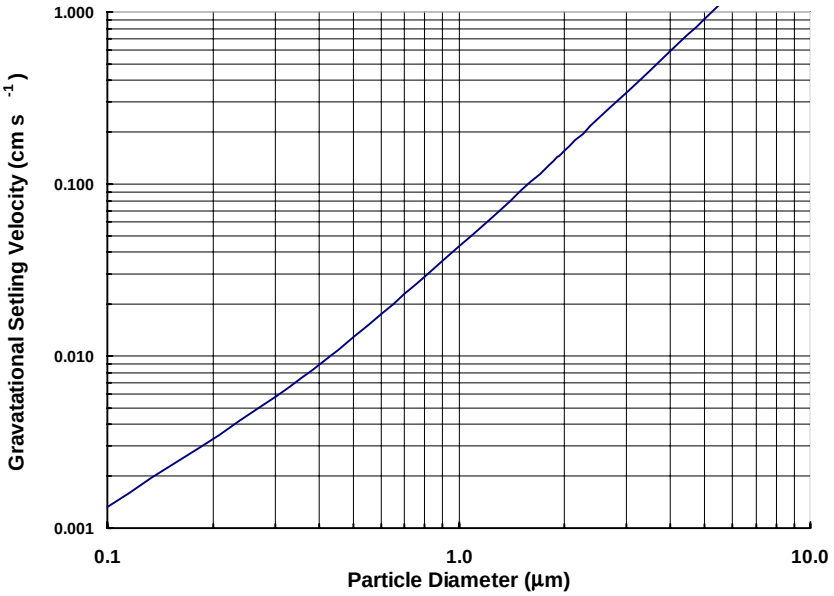

Figure 5. Gravitational settling velocity as a function of particle diameter for plutonium (ρ = 11.46 g cm^{-3}).

Calculations with RATCHET showed that the average difference among airborne concentrations at the 15 selected receptor locations in the model domain for 1 and 10 µm AED particles was a factor of 1.6. Therefore, particle size and subsequent deposition and plume depletion may not be as important as other sources of uncertainty in terms of estimating airborne concentrations, deposition, and cancer risk (However, the risk coefficients for 1- and 10-µm

particles are quite different.). For example, the uncertainty in predicting airborne concentrations from an inert tracer is roughly an order of magnitude, as revealed by the model comparison study previously discussed (Rood 1999a). Therefore, we did not anticipate gravitational settling would make a significant difference in the overall dispersion estimates, but we included the process in our calculations nevertheless. Although we assumed all particles were respirable (<10 μm AED), because we lack definitive data on the particle size, we treated this parameter as uncertain and assigned a distribution having a GM of 5 μm AED with a GSD of 2.5, and we truncated the distribution at 0.1 μm and 10 μm AED. This is not to say we assumed a particle size distribution of released plutonium as described by a GM of 5 μm and a GSD of 2.5. Rather, the particle size on which gravitational settling was calculated for each Monte Carlo trial was selected from a distribution having a GM of 5 μm and a GSD of 2.5.

Wet deposition of small particles in RATCHET is modeled using a washout coefficient and assuming irreversible collection of particles as the precipitation falls through the puffs. Precipitation was not recorded in the model domain during the 1957 fire release event; therefore, this process is irrelevant.

Diffusion Coefficients

In RATCHET, the diffusion coefficients are estimated directly from statistics for atmospheric turbulence. In most cases, the statistics describing atmospheric turbulence (i.e., standard deviation of the horizontal and vertical wind direction fluctuations) are not routinely measured at most meteorological recording stations. However, RATCHET makes use of atmospheric conditions that are either measured or calculated from routine meteorological data to estimate the turbulence statistics. The parameters wind speed, atmospheric stability, and surface roughness are used to estimate the turbulence statistics. The general form of the equation used in RATCHET for estimating the horizontal diffusion coefficient (σ_r) for the first hour following release is

$$\sigma_r = 0.5\sigma_v\, t \tag{7}$$

where

σ_v = crosswind component of turbulence (m s^{-1})

t = travel time.

After the first hour, the horizontal diffusion coefficient is given by $\sigma_r = c_{sy}\, t$, where c_{sy} is a proportionality constant with dimensions of meters per second. Gifford (1983) has shown the value of c_{sy} distributed between 0.14 to 1.4 with a median value of 0.5. For our simulations, we used the median value of 0.5.

The general form of the equation for estimating the vertical diffusion coefficient (σ_z) near the source is

$$\sigma_z = \sigma_w\, t\, f_z(t) \tag{8}$$

where

σ_w = standard deviation of the vertical component of the wind (m s^{-1})

$f_z(t)$ = nondimensional function related to the travel time and turbulence time scale.

As a practical matter, diffusion coefficients in RATCHET are calculated in increments to avoid problems associated with spatial and temporal changes in conditions.

The RATCHET documentation states that the diffusion coefficients implemented in the code are not appropriate for instantaneous puffs. Puff diffusion is defined as when the sampling time is short compared to the travel time of the airborne material (Hanna et al. 1982). Travel times to the eastern margin of the model domain were ~3 hours. If we were interested in the instantaneous concentration at points in the model domain, then puff diffusion coefficients would be required for the simulation. However, we are not calculating instantaneous concentrations, rather *TICs* over the assessment period (9 hours). Because our sampling time (9 hours) is greater than the travel time, the plume diffusion coefficients discussed in this section are appropriate for the simulation.

Source Characterization

Release estimates of plutonium particles that were <10 μm to the atmosphere were provided by Voillequé (1999a) and are summarized in a previous section or this report. Release estimates were segregated into 15-minute time intervals starting at 10:00 p.m. on September 11, 1957, and continuing until 2:00 a.m. the following day. For each 15-minute time interval, a nonparametric distribution of release quantities was provided. The distribution was described in terms of percentiles in 5% increments. Originally, RATCHET allowed hourly source updates. The code was modified to allow source updates for every new puff introduced into the model domain. The number of puffs per hour was set to four so source updates would be recorded every 15 minutes. Each source update consisted of the *quantity* of plutonium released during the source update time increment (15 minutes). Monte Carlo simulations were performed by selecting two random numbers at the beginning of each trial. These random numbers were used to select a percentile from the source term distributions. The same percentile was used for each time interval within the trial, resulting in source release rates that varied only by the total amount of plutonium released during the event (Figure 6).

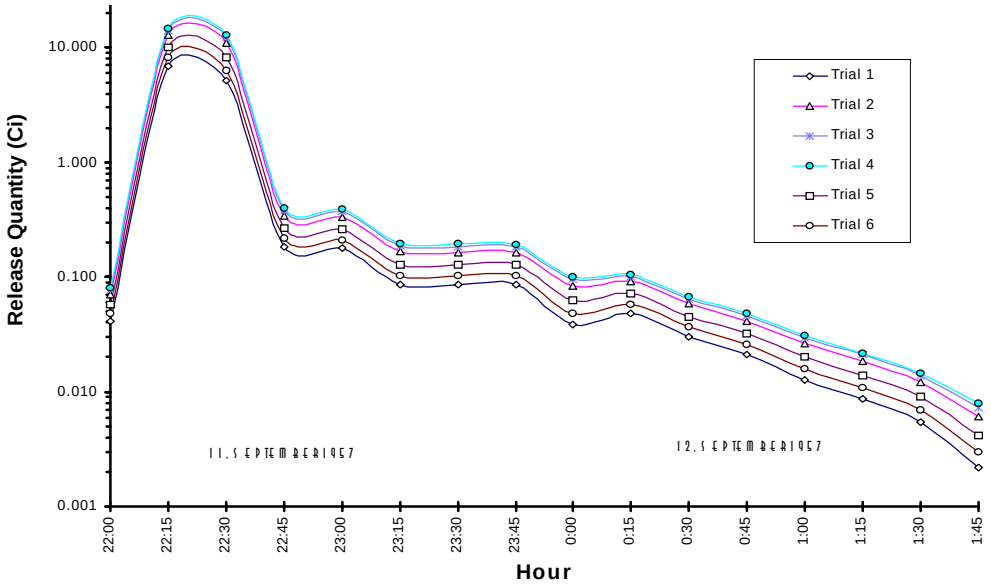

Figure 6. Release from the Building 771 stack as a function of time for six Monte Carlo trials. For each trial, the timing of the release is not changed; only the total quantity of plutonium released is changed.

Release quantities were provided until 2:00 a.m. September 12, 1957. After that, the source was assumed to go to zero and transport calculations were continued until 7:00 a.m. to allow all contaminant mass in the model domain to dissipate. Therefore, the RATCHET simulations were performed for 9-hours.

Suspended plutonium resulting from the glove box fire was vented through the 44-m Building 771 stack (Table 5). Initially, the stack was operating at its standard design flow rate of 85 m^3 s^{-1} (velocity of 11.6 m s^{-1}). After the explosion in the ventilation system and subsequent fire in the filter bank (10:39 p.m.), the main exhaust fans were shut down. Voillequé (1999a) estimates the volume of combustion products from the fire resulted in a stack flow rate between 13,000 and 14,000 ft^3 m^{-1} (6.37 m^3 s^{-1}). Temperature of the effluent at the stack after 10:40 p.m. was estimated to be 404°C (760°F). The highest releases occurred during the period from 10:15 p.m. to 10:45 p.m. and were several orders of magnitude higher than releases before the explosion. Plume rise was calculated internally by RATCHET, and the methods used are described below.

Plume Rise. Momentum and buoyant plume rise is calculated in RATCHET using the equations proposed by Briggs (1969, 1975, 1984). The implementation of these equations in RATCHET was taken from the INPUFF2 model (Petersen and Lavdas 1986) and is a function of the wind speed at the release height, stability class, release temperature, ambient temperature, and the stack flow rate. The following description was taken from the RATCHET documentation (Ramsdell et al. 1994).

Plume rise is caused by two factors, vertical momentum of the exhaust gases in the stack and buoyancy because of the density differences between the stack gases and the atmosphere. In general, one factor or the other dominates the overall process. For a given set of stack and

atmospheric conditions, the temperature difference between the stack effluent and the air determines which of the factors is dominant. The plume rise methodology calculates a critical temperature difference that separates two regimes (momentum or buoyancy driven). When the actual temperature difference is less than the critical temperature, momentum is the dominant factor determining plume rise. Otherwise, plume rise is driven by buoyant forces. Effluent temperatures in Table 5 suggest plume rise was dominated by momentum forces before and during the early stages of the fire (before 10:40 p.m.) and buoyant forces after the exhaust fans were shut off.

Under unstable and neutral atmospheric conditions, plume rise is dominated by momentum as long as the temperature difference between the plume and the air is less than a critical temperature difference given by

$$\Delta t_c = C\, wp^{1/3}\, Tp\left(2\, r_s\right)^{-2/3} \tag{9}$$

where

Δt_c = critical temperature difference (K)
C = dimensional constant which arises from the combinations of equations 10, 11, 12, and 13 are solved for Δt_c; assuming the use of metric units, the constant is 0.0297 $(\text{m s})^{1/3}$
wp = stack exit vertical velocity (m s^{-1})
Tp = initial plume temperature (°K)
r_s = inside stack radius (m).

When the difference between initial plume temperature (*Tp*) and the ambient temperature (*Ta*) is less than Δt_c, plume rise is estimated using

$$\Delta h = 6 r_s \frac{wp}{u(h_s)} + \Delta hd \tag{10}$$

where

Δh = plume rise above the initial release height (m)
$u(h_s)$ = wind speed at the initial release height (m s^{-1})
Δhd = downwash correction (m).

The downwash correction is applied if the stack exit velocity is less than 1.5 times the wind speed at the release height and is given by

$$\Delta hd = 4 r_s \frac{wp}{u(h_s)} - 1.5\ . \tag{11}$$

If *Tp* – *Ta* is greater than Δt_c, the plume rise is estimated using the equation for buoyancy-dominated rise given by

$$\Delta h = 1.6 Fb^{1/3}\, xf^{2/3}\, u(h_s)^{-1} + \Delta h_d \tag{12}$$

where F_b = the buoyancy flux parameter ($\text{m}^4\ \text{s}^{-3}$) and *xf* = the distance to final plume rise (m). The buoyancy flux parameter is given by

$$F_b = g\left(\frac{Tp - Ta}{Tp}\right) wp\ r_s^2 \tag{13}$$

where g = gravitational acceleration (9.8 m s^{-2}). According to Petersen and Lavdas (1986), the distance to final plume rise is given by

$$xf = 49\, Fb^{5/8} \tag{14}$$

where the leading constant (49) has units of $s^{15/8}$ $m^{3/2}$.

Under stable atmospheric conditions, the critical temperature difference is given by

$$\Delta t_c = 0.0196\, wp\ \ Ta\ \ S^{1/2} \tag{15}$$

where S = the stability parameter and the dimensions of the constant (0.0196) are m s^{-1}. The parameter S is computed from the stability class and air temperature from

$$S = g\, Ta \frac{\partial \theta}{\partial z} \tag{16}$$

where $\partial\theta/\partial z$ = the potential temperature lapse rate. Potential temperature lapse rates of 0.02 K m^{-1}, 0.035 °K m^{-1}, 0.05 K m^{-1} are assumed in RATCHET for stability class 5, 6, and 7, respectively. When $Tp - Ta$ is less than Δt_c, momentum-dominated plume rise is estimated using Equation (11). It is also estimated using

$$\Delta h = 1.5 S^{1/6} \left(\frac{F_o\, wp\, Ta}{\pi\, u(h_s)\, Tp} \right)^{1/3} + \Delta hd \tag{17}$$

where F_o = the stack flow rate in m^3 s^{-1}. The final estimate used in RATCHET is the smaller of these two values. When Tp – Ta is greater than Δt_c, one of two equations is used to estimate plume rise. If the wind speed at the release height is greater than a critical wind speed, U_c, defined by

$$U_c = 0.275 F_b^{1/4}\ \ S^{1/8} \tag{18}$$

then the plume rise is calculated using

$$\Delta h = 2.6 F_b^{1/3} \left(S\, u(h_s) \right)^{-1/3} + \Delta hd \quad . \tag{19}$$

If the wind speed at the release height is less than U_c during stable conditions, the plume rise is computed using

$$\Delta h = 4 F_b^{1/4}\ S^{-3/8} + \Delta hd \quad . \tag{20}$$

The wind speed at the release height is given by

$$U(z) = \frac{u_*}{k} \left(\ln(z/z_o) - \psi(z/L) \right) \quad . \tag{21}$$

where u_* = the friction velocity (m s^{-1}), k = the von Karman constant (0.4), ψ = the stability correction factor, and L = Monin-Obukhov length (m).

Plume rise was calculated for the first hour of the release for three different cases. The first case assumed the operating flow rate and temperature (85.5 m^3 s^{-1} and 20°C, respectively) that would have been present during the initial stages of the fire. The second case assumed the operating flow rate of 85.5 m^3 s^{-1} and an effluent temperature of 200°C, representing the

conditions from about 10:20 p.m. to 10:40 p.m., September 11 during the early stages of the fire. The third case assumed release conditions after the explosion and loss of power to the exhaust fans (flow rate of 6.37 m^3 s^{-1} and release temperature 404°C). Calculated plume height above the stack for the three cases were 43.8 m, 73 m, and 46.5 m respectively. Note that plume rise is not that much different between cases 1 and 3. Because these transitions occurred during first hour of the release, and that uncertainty exists in wind speed and direction estimates aloft, we have chosen to use releases conditions represented by the third case for the entire release. In doing so, we may have underestimated the plume height reached during some of the release. If such is the case, then air concentrations will likely be overestimates because a greater plume height will result in greater mixing and dispersion of airborne material. As eluded to in an earlier section, if the plume penetrated the surface-layer air mass and was entrained an upper-level air mass, then we would have little hope in predicting its trajectory because no meteorological measurements exist for upper air levels for the 1957 fire. Release parameters used in the RATCHET simulations are summarized in Table 5. Hourly ambient temperatures at the 10-m level were used in the RATCHET simulation (Appendix A).

Table 5. Release Parameters for Building 771 Stack

Parameter	Value
Stack height	44 m
Stack diameter	3.05 m
Stack flow rate before 10:39 p.m.[a]	85.5 m^3 s^{-1}
Stack flow rate after 10:39 p.m.[b,c]	6.4 m^3 s^{-1}
Effluent temperature before 10:20 p.m.[a]	20°C
Effluent temperature 10:20 p.m. – 10:39 p.m.	200°C
Effluent temperature after 10:39 p.m.[b,c]	404°C
UTM east	482,947 m
UTM north	4,415,997 m

a. Normal operating conditions
b. Estimated temperature and flow rate of the effluent after the fire spread to the filter bank and power was cut to the exhaust fans.
c. Transport calculations used the post-filter bank fire conditions for the entire release event (i.e. a temperature of 404° C and flow rate of 6.4 m^3 s^{-1}.

Other Parameters

Several other parameters in RATCHET influence the accuracy of output and computer runtime. These parameters include the number of puffs per hour, minimum time step, puff consolidation, maximum puff radius, and minimum puff concentration at center. We chose the suggested RATCHET default values for all these parameters except minimum time step and minimum concentration at puff centers (Table 6). Accuracy of the simulation can be improved by using a smaller time step. The RATCHET default was 20 minutes, which we reduced to 10 minutes. The minimum concentration at puff centers was reduced from 1×10^{-13} to 1×10^{-15} to allow for plume tracking throughout the model domain. The puff consolidation parameter value combines puffs from the same source when the ratio of the puff centers to the average σ_r is less than the user-input value. The puff consolidation ratio and maximum puff radius (in units of σ_r) were set at RATCHET default values of 1.5 and 3.72, respectively.

Table 6. RATCHET Model Control Parameters

Model parameter	Value
Number of puffs per hour	4
Minimum time step	1 minute
Puff consolidation	1.5
Maximum puff radius (in units of σ_r)	3.72
Minimum concentration at puff centers	1×10^{-15}

Prediction Uncertainty

The uncertainty analysis for the 1957 fire dispersion estimates employed the random sampling features of the RATCHET code because (1) meteorological data were available for the specific release event and (2) random sampling of meteorological input parameters allows for mass balance of the source term with the contaminant mass in the model domain. RATCHET uses random sampling from specified distributions to represent the uncertainty in meteorological data. Specifically, random sampling is limited to wind directions and wind speeds, stability class, Monin-Obukhov length, precipitation rates, and station mixing layer depths. This limitation preserves the physically based correlations among other model parameters and variables. Random sampling of precipitation rates was not used because precipitation did not fall during the event. Uncertainty in the source term and particle size was handled external to the RATCHET code.

Wind Direction Uncertainty. Uncertainty in the wind direction is addressed in RATCHET by sampling from a uniform distribution whose width depends on the measured wind speed. During calm conditions, the width of the distribution is from 0 to 360 degrees. The distribution narrows as the wind speed increases, until the width of the distribution equals the imprecision in the recorded values (a minimum value of 10 degrees). The method used to vary the width of the distribution in RATCHET is based on a procedure described in Schere and Coates (1992). Other sources of uncertainty in wind directions are not considered by the random-sampling algorithm in RATCHET. These sources of uncertainty include

- Instrument exposures that may cause observed wind direction to differ systematically from the directions that are representative for the region of measurement
- Changes in wind direction with height that may cause elevated plumes to move in a direction that is different from the one predicted from surface observations.

In reference to the last bullet, Elderkin and Gudiksen (1993) studied several of the WVTS nighttime tests in which additional instrumentation was installed and monitored as part of the Atmospheric Studies in Complex Terrain (ASCOT) program. They found dispersion was controlled by multiple scales of motion, which created interacting layers that varied hourly in three dimensions. Tracer plumes were mostly confined to a stable drainage layer that followed regional flow features, intermittently interrupted by evolving mountain-canyon flows. Based on conventional surface observations, interactions between the surface layer and the mountain-canyon flow layer caused unexpected tracer trajectories. In all atmospheric model simulations performed for this project, we assumed the contaminant plumes remained confined to the surface layers. We acknowledge the possibility that some of the plumes may have been entrained in upper layers as described by Elderkin and Gudiksen (1993). However, we find it impossible to

predict the trajectory of such entrained material with any confidence because of the lack of meteorological data and the current state-of-the art of atmospheric transport models. If plumes were entrained in upper layers, ground-level plutonium air concentrations in the model domain would decrease because contaminant mass would remain aloft.

Wind Speed Uncertainty. Wind speeds are recorded in some meteorological records as integer values and in a variety of units. For example, the Denver Stapleton International Airport wind speeds are reported to the nearest whole number in units of knots. This imprecision in wind speed measurements is addressed in RATCHET. RATCHET also addresses the additional uncertainty in wind speeds near and below the threshold.

When random sampling of wind speeds is selected, wind speed is drawn from a uniform probability distribution because with a given wind observation there is no reason to assume that the actual speed is more or less likely to be in any part of the range of values. The width of the distribution is two reporting units. For example, if the measured wind speed is 5 m s^{-1}, then the width of the distribution is from 4 to 6 m s^{-1}. When a calm wind is reported, a wind speed between 0 and 1 m s^{-1} is used.

Stability Uncertainty. Atmospheric stability is a fundamental concept in meteorology, but it cannot be calculated directly from the available meteorological data. Therefore, stability must be estimated from the limited data that are available.

Methods of estimating stability classes proposed by Gifford (1961), Pasquill (1961), and Turner (1964) are based on data that are available in routine meteorological observations, such as those taken at airports. These methods form the basis of the procedures that the National Climatic Data Center uses to estimate stability classes from climatological data (Hatch 1988).

Golder (1972) compares stability class estimates made at five locations using the method proposed by Pasquill and Turner's variation. The results of this comparison, presented in Golder (1972, Figure 3), show reasonable agreement among the hourly stability-class estimates. However, other studies, such as the study of Luna and Church (1972), show that these stability classes have a much wider range of uncertainty when attempting to estimate turbulence characteristics related to diffusion.

RATCHET allows the user to specify the uncertainty associated with stability class estimates. This uncertainty is represented by a set of seven conditional cumulative frequency distributions—one conditional cumulative frequency distribution for each stability class. The cumulative frequency distribution represents the possible actual stability class for the one reported stability class. To do this, two different methods of calculating stability were employed: (1) the method described in Turner (1964) and used to define nominal values for stability class and (2) the lateral turbulence and wind speed method (standard deviation of the horizontal wind direction fluctuations) as described in EPA (1987). Stability classes were calculated for 5 years of meteorological data taken at the RFP between 1989 and 1993. This was the same meteorological data set used for routine release and transport calculations (Rood 1999b). Conditional cumulative frequency distributions were input through a file containing 7 records, one for each stability class (Table 7). Each record contains seven values that are the cumulative probability that the actual stability class is the same as the reported stability class.

For example, the probability that a reported stability class of 1 is actually 1 is 0.934 (see Table 7, line one column 2). The probability that a reported stability class of 1 may actually be 2 is 0.961 – 0.934 = 0.027. The probability that a reported stability class of 1 may actually be 3 is

0.988 – 0.961 = 0.027. The probability that a reported stability class of 1 may actually be 4 is 1.00 – 0.988 = 0.012 and so on.

Table 7. Conditional Cumulative Frequency Distributions for Stability Class

Stability class	Cumulative frequency that the actual stability class is ≤ the reported stability class						
	1	2	3	4	5	6	7
1	0.934	0.961	0.988	1.000	1.000	1.000	1.000
2	0.565	0.819	0.927	0.996	0.998	1.000	1.000
3	0.268	0.409	0.704	0.955	0.981	1.000	1.000
4	0.072	0.113	0.213	0.895	0.975	1.000	1.000
5	0.000	0.000	0.000	0.597	0.994	1.000	1.000
6	0.000	0.000	0.000	0.339	0.629	1.000	1.000
7	0.000	0.000	0.000	0.110	0.279	1.000	1.000

Monin-Obukhov Length. Stability classes are discrete estimates of atmospheric stability. However, boundary-layer similarity theory uses the reciprocal of the Monin-Obukhov length, which is a continuous variable, to represent stability. Figure 2.4 in the RATCHET documentation (Ramsdell et al. 1994), which is based on Golder's (1972) paper, provides a basis for converting stability class to Monin-Obukhov length (Figure 7). When random sampling of the reciprocal Monin-Obukhov is selected, RATCHET obtains an appropriate value as needed from a uniformly distributed range of values. The upper and lower bounds of the range are computed from the surface roughness and stability class.

Mixing Layer Depth. RATCHET computes mixing-layer depth from the friction velocity and Monin-Obukhov length. For stable conditions, the mixing layer depth is given by

$$H = k\left(\frac{u_* L}{f}\right)^{1/2} \tag{22}$$

where

H = mixing layer depth (m)
k = von Karman constant (~0.4 dimensionless)
u_* = friction velocity (m s^{-1})
L = Monin-Obukhov length (m)
f = Coriolis parameter (s^{-1}).

Pasquill and Smith (1983) indicate that constant values in the range of 0.2 to 0.7 have been suggested in place of the von Karman constant. Weil (1985) suggests constant values in the range of 0.4 to 0.7. For neutral or unstable conditions, the mixing layer depth is estimated using

$$H = \frac{\beta u_*}{f} \tag{23}$$

where β = a dimensionless constant. Zilitinkevich (1972) assumes β is equal to k, while Pasquill and Smith (1983) suggests a value in the 0.2 to 0.3 range. Other researchers (Panofsky and Dutton 1984) suggest its range is from 0.15 to 0.25. When random sampling of the mixing depth is selected, RATCHET samples from uniform distributions the value of k and β. The range of k is fixed between 0.2 and 0.7 and the range of β is fixed between 0.15 and 0.3.

Integration of Uncertainty Analysis into Model Predictions. The uncertainty analysis required a Monte-Carlo simulation that coupled RATCHET atmospheric transport simulations with distributions of the 1957 fire source term. Calculations were performed using a FORTRAN pre-and post-processor program that (1) sampled release rate and particle size from distributions of these quantities, (2) wrote RATCHET input files, (3) executed the RATCHET simulation, and (4) extracted and stored results. The source release rate and meteorological parameters were considered independent of one another.

Output consisted of 1000 trials of 9-hour average atmospheric concentrations and deposition at the 2295 receptor nodes in the model domain. Average concentrations were converted to *TICs* by multiplying by the number of hours in the simulation (9 hours). The TIC values were used in the intake and risk calculations.

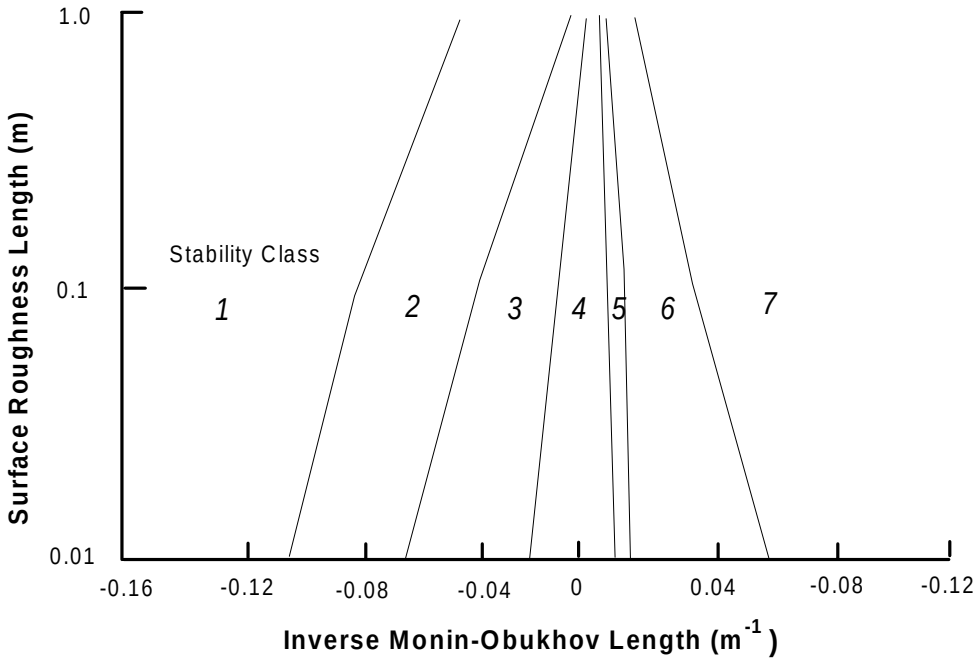

Figure 7. Relationship between stability class and Monin-Obukhov length as a function of surface roughness length (redrawn from Ramsdell et al. 1994).

Predicted Air Concentrations

Average and time-integrated plutonium concentrations were calculated throughout the model domain using the source term developed by Voillequé (1999a) and the atmospheric modeling procedure described in previous sections. Distributions of 9-hour time-integrated air concentrations for 15 receptor locations have been described in terms of the 5th, 50th, and 95th percentile values of the distribution of predicted air concentration values (Table 8). We used these statistics to describe the concentration distributions; they were not used in the risk calculation. Instead, we used the actual distributions comprising 1000 RATCHET realizations to calculate plutonium intake and risk to the receptors. The 15 receptor locations chosen for risk

calculations represented individuals from each of the major population centers in addition to receptors placed at locations of high concentration in the model domain. For many of the receptor locations in Table 8, the 5th percentile value was zero, indicating in 5% of the 1000 transport simulations, the plume did not reach the receptor over the 9-hour simulation period. Consequently, it is difficult to express concentrations in terms of lognormal statistics such as GM and GSD. The maximum time-integrated air concentration at the 95% level for the 15 receptor locations was 3600 pCi-h m^{-3} and occurred south of the RFP along the buffer zone boundary (Table 8). The receptor in Boulder had zero concentration at the 95% value, indicating a low probability based on the modeling that the plume ever reached that location during the period simulated (10:00 p.m. September 11 to 7:00 a.m. September 12).

Time-integrated concentrations in the entire model domain ranged from a minimum of 0 to a maximum of 14 pCi-h m^{-3} at the 5% level and 0 to 3600 pCi-h m^{-3} at the 95% level. Figure 8 shows the number of simulations where the 9-hour average concentration was greater than zero for 1000 Monte Carlo trials. The 95% contour line (950 of the simulations had a 9-hour average concentration greater than zero) encompassed an area that includes the cities of Arvada, Westminster, Federal Heights, Thornton, and Northglenn. Unlike the routine release evaluation (Rood 1999b) where nonzero concentrations were calculated for all receptor nodes, this event has an additional component of uncertainty; that is, the probability that the plume even reached the receptor.

The median estimate (50th percentile value) of the 9-hour average concentration is illustrated in Figure 9. The dispersion pattern depicted represents the 50th percentile concentration estimate at each of the 2295 receptor nodes. Dispersion patterns are typical of what we would expect for an elevated release from the RFP. The plume is initially lofted and does not touch down until reaching an area north of Arvada. The plume trends southeast from the RFP until it reaches the western margin of the Platte River Valley, located about 10–12 km southeast of the RFP. At that point, air movement down the Platte River Valley, which is typical of evening conditions, causes the plume to move in a northeasterly direction. This trend is believed to be due to the influence of the air movement up and down the Platte River Valley and the diurnal pattern of upslope-downslope conditions that characterize the general air movement on the Colorado Front Range environs (Crow 1974). Downslope conditions typically occur during the evening hours and are characterized by drainage flow of cooler air from the foothills to the plains. Westerly winds predominate, but the direction may be altered by local topography and conditions. Upslope conditions are a result of daytime heating and typically result in easterly winds that prevail during the daylight hours. The transition from upslope to downslope conditions occurs during the evening, and transition from downslope to upslope occurs during the morning. During evening hours under stable conditions, cool air near the surface drains from the Denver metropolitan area down the Platte River Valley (which flows to the northeast) and out to the plains. During daylight hours and after surface heating has eliminated the cooler surface layer, the downslope conditions cease. This is followed by a brief period of relatively calm winds, which in turn is followed by return of air up the valley or upslope conditions.

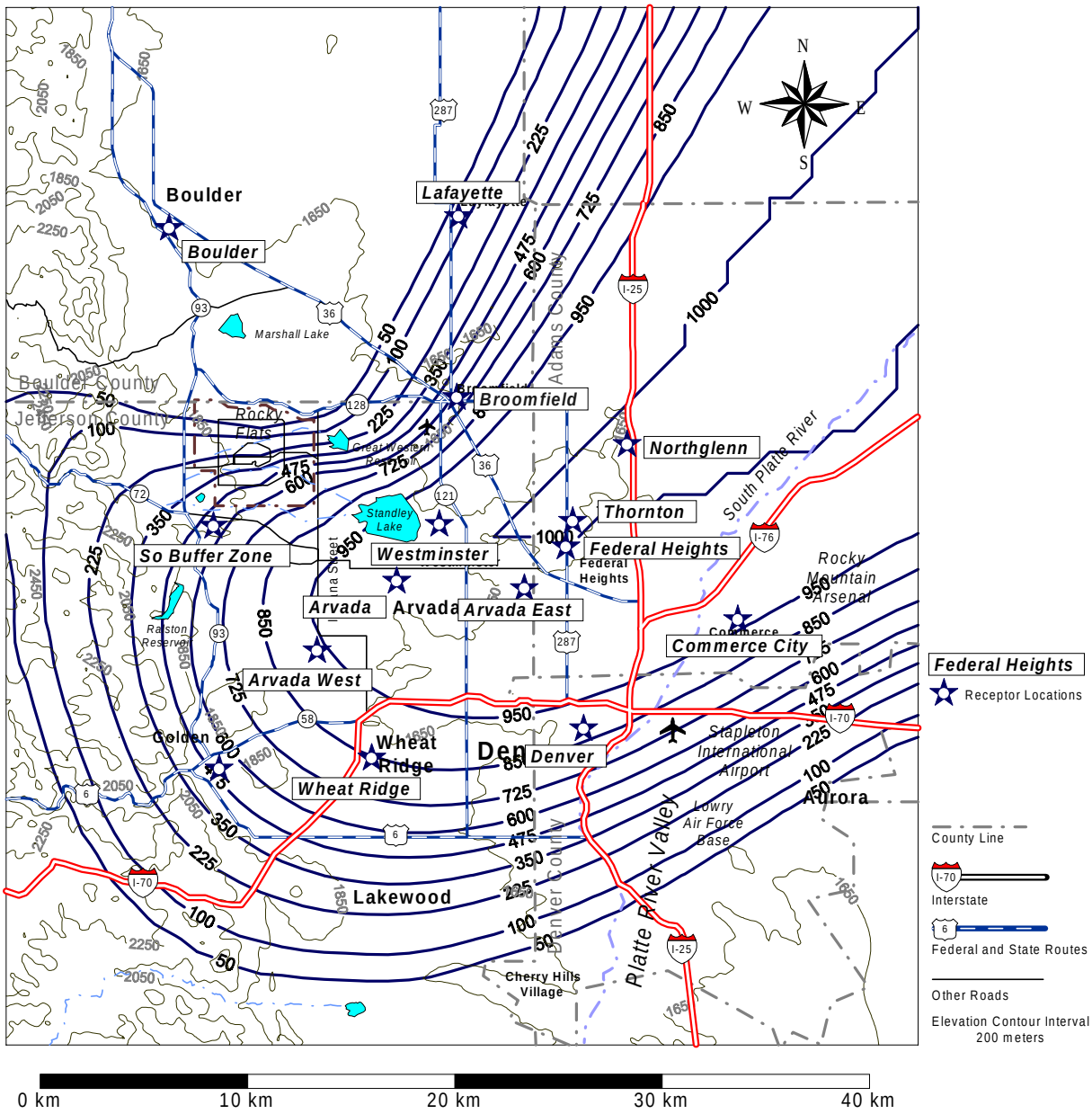

Figure 8. Number of Monte Carlo trials where the 9-hour average plutonium concentration in air was greater than 0 for 1000 trials. A value of 500 represents a 50% probability that the plume passed over that location. Receptor locations used in the risk calculations are indicated by a star. The receptor location name is indicated by the text within the rectangle.

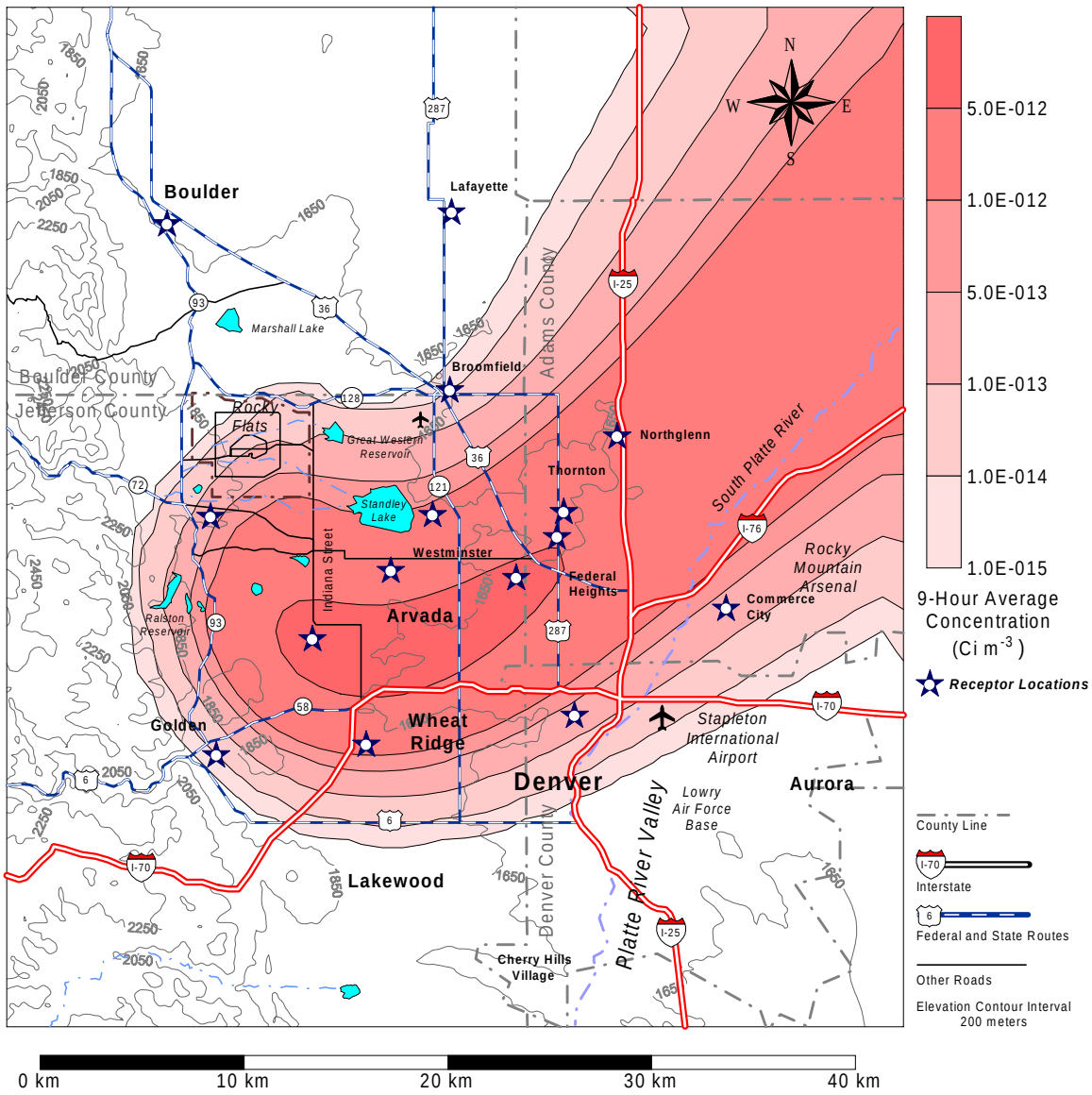

Figure 9. Predicted 9-hour average plutonium concentration in air at the 50th percentile level. Concentrations were based on 1000 Monte Carlo trials. Receptor locations that are used in the risk calculations are indicated by a star.

Table 8. Predicted Time-Integrated Concentrations for Plutonium in Air at 15 Receptor Locations in the Model Domain

Receptor location			*TIC* (pCi-h m^{-3})		
UTM E (m)	UTM N (m)	Description	5%	50%	95%
495850	4409050	Arvada East	1.4×10^{1}	5.8×10^{1}	1.9×10^{2}
497850	4411050	Federal Heights	1.3×10^{1}	4.9×10^{1}	1.5×10^{2}
485850	4406050	Arvada West	0.0×10^{0}	7.2×10^{1}	7.7×10^{2}
480850	4412050	So RFP Buffer	0.0×10^{0}	5.6×10^{-1}	3.6×10^{3}
489696	4409400	Arvada	1.7×10^{0}	2.4×10^{1}	4.1×10^{2}
488491	4400870	Wheat Ridge	0.0×10^{0}	4.7×10^{0}	1.8×10^{2}
491747	4412140	Westminster	4.9×10^{-1}	9.9×10^{0}	7.3×10^{1}
492593	4418250	Broomfield	0.0×10^{0}	3.8×10^{-4}	5.7×10^{-1}
500804	4415990	Thornton	6.9×10^{0}	2.9×10^{1}	8.3×10^{1}
498184	4412290	Northglenn	1.1×10^{1}	4.1×10^{1}	1.5×10^{2}
478709	4426410	Boulder	0.0×10^{0}	0.0×10^{0}	0.0×10^{0}
492674	4426980	Lafayette	0.0×10^{0}	0.0×10^{0}	1.0×10^{-4}
481126	4400350	Golden	0.0×10^{0}	0.0×10^{0}	7.1×10^{1}
506152	4407550	Commerce City	5.2×10^{4}	1.5×10^{0}	1.6×10^{1}
498712	4402290	Denver	0.0×10^{0}	2.6×10^{-1}	1.4×10^{1}

The initial plume trajectory is controlled by conditions at Rocky Flats that, at the time of release, transported the plume in a westerly direction for a few kilometers. Wind direction shifted around 10:45 p.m. to out of the northwest and continued to blow from that direction until about 4:00 a.m. September 12. These winds transported the bulk of the airborne plutonium to Arvada and toward the Denver metropolitan area. Near southern Arvada, the air mass converged with southwest air flow in the Platte River Valley resulting in northeasterly trending plume trajectory.

Recall that two plumes were identified in Phase I of this study: one trending southeast and one trending more-or-less easterly. There are two reasons why we did not observe this phenomenon in our simulations. First, the majority of the Phase II release was postulated to occur over a relatively short period of time (1-hour); in Phase I, the release was assumed to occur over a longer period of time. The Phase I source assumed a release rate of 4 μCi s^{-1} from 10:40 p.m. September 11 to 2:00 a.m. September 12th. This was followed by a release rate of 0.07 μCi s^{-1} from 2:00 a.m. to 11:30 a.m on the morning of September 12th. During the period the release rate was 4 μCi s^{-1}, the wind changed direction from out of the northwest (11:00 p.m. to midnight) to more-or-less westerly (midnight to 2:00 a.m. September 12th), resulting in two plumes. Second, incorporating Denver Stapleton meteorological data into the Phase II simulation forced the plume down the Platte River Valley, thereby, changing its original trajectory defined by meteorological conditions at Rocky Flats.

EXPOSURE SCENARIOS AND RISK CALCULATIONS

One of the key parts of the Rocky Flats dose reconstruction work is calculating health impacts to people living in the surrounding area from materials released during RFP past operations. Dose reconstruction uses a pathways approach to study the potential radiation doses

and health risks of these past releases on the surrounding communities. The pathways approach begins with learning what kinds of and how much materials were released from a facility (source terms) and ends with estimating the health impacts these releases had on the residents in the area. Mathematical models described in the previous sections were used to model the transport of plutonium released from the site to the surrounding communities. In this section, we present the method for calculating health impacts (incremental lifetime cancer incidence risk) to people living offsite from exposure to these releases.

The risk to a person from exposure to the plutonium released depends upon a number of factors, such as

- Where the person lived and worked in relation to the RFP
- Did the person live near the RFP during the 1957 fire
- The age and gender of the person
- Lifestyle (that is, did the person spend a great deal of time outdoors or doing heavy work on a farm).

Although it is not realistic to calculate individual risks for every resident who may have lived or worked in the Rocky Flats area during its operational history, it is not credible to calculate a single risk that applies to all residents. To consider the many factors that influence exposure, we developed profiles, or exposure scenarios, of hypothetical, but realistic residents of the RFP area for which representative risk estimates could be made. Each scenario represents one individual. These scenarios incorporate typical lifestyles, ages, genders, and lengths of time in the area. The scenarios also specify the home and work locations. These scenarios can help individuals determine risk ranges for themselves by finding a lifestyle profile that most closely matches their background. The scenarios are not designed to include all conceivable lifestyles of residents who lived in this region during the time of RFP operations. Rather, they provide a range of potential profiles of people in the area.

We calculated the risks to hypothetical individuals from plutonium released to the air during the 1957 fire for seven hypothetical exposure scenarios (Table 9) at 15 separate locations in the model domain. Locations were selected to include all major population centers and to intercept the plume path where the maximum concentration occurred. As discussed earlier, direct inhalation was the only exposure pathway considered in this assessment. Ingestion of plutonium in water, food, and soil are potential pathways that could have been considered in more detail. However, plutonium compounds are very insoluble and tend to adhere to soil, making them relatively immobile and not readily taken up by plants or accumulated in the edible portions of animal products. Phase I results (ChemRisk 1994d) indicated direct inhalation to be the dominant pathway of exposure during the early period of RFP operations (1952–1970). For the later years (1970–1989), soil ingestion and inhalation of resuspended contaminated soil become a significant component of the total dose because of the accumulation and build up of deposited plutonium in soil and smaller airborne releases. This report deals only with risks from the 1957 fire, which are dominated by direct inhalation of airborne activity released by the fire. Risk from inhalation of resuspended soil activity will be addressed in a later, more comprehensive report that addresses all sources of offsite plutonium contamination.

Table 9. Exposure Scenario Descriptions

Exposure scenario	Gender	Year of birth	Locations
Rancher	Male	1925	South of RFP at the buffer one fence, Arvada West
Office worker	Female	1941	Denver
Housewife	Female	1928	Arvada, Wheat Ridge, Westminster, and Broomfield
Laborer	Male	1943	Thornton, Boulder, and Lafayette, Arvada East, Federal Heights
Infant	Female	1957	Northglenn
Child	Female	1953	Commerce City
Student	Female	1949	Golden

Exposure scenarios for the seven hypothetical receptors described in Table 9 were organized according to occupational and nonoccupational activities. Occupational activities included work, school, and extracurricular activities away from the home. Nonoccupational activities included time spent at home doing chores, sleeping, and leisure activities (such as watching television). In these calculations, the receptor was assumed to perform occupational and nonoccupational activities at the same location. The age of the receptor during which exposure occurred was also considered when calculating risk. All scenarios assumed the individual was exposed for the duration of the 1957 fire event.

Breathing Rates and Time Budgets

Each exposure scenario was divided into three types of activities: sleeping, nonoccupational, and occupational activities. For the infant and child scenario, occupational and nonoccupational activities are irrelevant; instead, activities were divided into sleeping and two other activities based on the child's age. For the infant, the other two activities were awake sedentary and awake active. For the child scenario, the two other activities were time spent at home (indoors and outdoors) and at preschool and/or day care.

For each activity, time spent at four different exercise levels was assigned. These exercise levels were resting, sitting (sedentary), light exercise, and heavy exercise. Some examples of light exercise are laboratory work, woodworking, housecleaning, and painting. Heavy exercise corresponds to occupations such as mining, construction, farming, and ranching. For each exercise level, an age- and gender-specific breathing rate was assigned. Breathing rates (Table 10) for persons age 8 and higher were obtained from Roy and Courtay (1991) and for children age 0–7 from Layton (1993).

Table 10. Breathing Rates for Various Exercise Levels as Reported in Roy and Courtay (1991) and Layton (1993)

		Exercise level			
Gender	Age	Resting ($m^3\ h^{-1}$)	Sitting ($m^3\ h^{-1}$)	Light ($m^3\ h^{-1}$)	Heavy ($m^3\ h^{-1}$)
Male	30–60	0.45	0.54	1.50	3.00
Female	30–60	0.32	0.39	1.26	2.70
Male	18	0.50	0.60	1.58	3.06
Female	18	0.35	0.42	1.32	1.44
Male	16	0.43	0.52	1.52	3.02
Female	16	0.35	0.42	1.30	2.70
Male	15	0.42	0.48	1.38	2.92
Female	15	0.35	0.40	1.30	2.57
Male	14	0.41	0.49	1.40	2.71
Female	14	0.33	0.40	1.20	2.52
Male	12	0.38	0.47	1.23	2.42
Female	12	0.33	0.39	1.13	2.17
Male	10	0.31	0.38	1.12	2.22
Female	10	0.31	0.38	1.12	1.84
Male	8	0.29	0.39	1.02	1.68
Female	8	0.29	0.39	1.02	1.68
Male	3–7	0.24	0.29	0.72	1.68
Female	3–7	0.23	0.27	0.68	1.59
Male	0–3	0.19	0.23	0.58	1.35
Female	0–3	0.14	0.17	0.45	1.02
Average, male[a]	8–17	0.37	0.45	1.28	1.49
Average, female[a]	8–17	0.33	0.40	1.18	2.25

[a] The average female breathing rate from age 8–17 was used for the student.

Time budgets for various receptor activities were also based on Roy and Courtay (1991) (Table 11), but they were modified to fit specific exposure scenarios and the timing of the fire event. The fraction of time spent at a specific exercise level while engaged in a given activity was assigned based on the nature of the activity. For example, the fraction of time spent at the resting exercise level while the receptor slept would be 1.0 and the other exercise levels would be 0. A weighted-average breathing rate was then applied to each activity based on the number of hours spent at each exercise level. For some scenarios (housewife, retiree, and laborer), nonoccupational activities were separated into those performed indoors and those performed outdoors. Although no distinction was made between indoor and outdoor air concentrations, exercise levels for indoor and outdoor activities differed. A time-weighted average breathing rate that included indoor and outdoor activities was calculated and applied to nonoccupational time.

Time-weighted average breathing rates were calculated for the three activities for which each receptor was assumed to be engaged. The time-weighted average breathing rate is given by

$$WBR_j = \sum_{i=1}^{4} BR_i \ f_{i,j} \qquad (24)$$

where
WBR_j = time-weighted average breathing rate for the j^{th} activity ($m^3\ h^{-1}$)
BR_i = breathing rate for the i^{th} exercise level ($m^3\ h^{-1}$)
$f_{i,j}$ = fraction of time spent at the i^{th} exercise level for the j^{th} activity.

To summarize, three activities were defined for each exposure scenario: sleeping, occupational, and nonoccupational activities. The location of exposure for occupational and nonoccupational activities was assumed to be the same for all receptors. Four different exercise levels, each with an assigned breathing rate, were distinguished: resting, sitting, light exercise, and heavy exercise. The breathing rate during a given activity was the time-weighted average breathing rate of the four exercise levels.

At the time of the major releases from the fire (10:00–11:00 p.m.), most people would be off work and either preparing to go to bed or in bed. Breathing rates while sleeping are substantially less than while awake (by about a factor of 3), and most people would have been sleeping for most of the release event. However, the major releases occurred over a period during which many people would have been awake or preparing for bedtime. Ideally, the calculation of plutonium intake would be described by

$$I = \int_0^t C(t)\, BR(t)\, dt \tag{25}$$

where $C(t)$ = the plutonium concentration as a function of time and $BR(t)$ = the breathing rate as a function of time. It was not practical to provide the function $C(t)$ for each Monte Carlo trial; instead, the *TIC* value was provided ($\int C(t)\, dt$). Therefore, the breathing rate applied to the intake calculation had to account for not only the activities performed by the receptor, but the fact that most of the intake occurred over the first several hours of the release event. Consequently, for the adult receptors, we assumed only 1 hour was spent at the sleeping breathing rate. We also assumed the remainder of the hours was spent doing occupational and nonoccupational activities because it is likely these receptors were awake during the major releases from the event. For the infant scenario, we assumed an equal amount of time for all three activity levels. For the child scenario, we assumed 6 hours of the exposure time was spent sleeping and the remainder was spent awake. For the student, who represents a person 7–18 years old, we assumed 3 of the 9 exposure hours were spent sleeping. The laborer scenario represents an individual who worked a graveyard shift; therefore, all his exposure time was spent engaged in occupational activities. This laborer scenario provides an upper-bound estimate of intake.

Plutonium Intake Calculation

The calculation of the incremental lifetime cancer incidence risk involved three steps:

1. Calculate the *TIC* in air at the point of exposure
2. Calculate the amount of plutonium inhaled by the receptor
3. Multiply the plutonium intake by a risk coefficient that relates the incremental lifetime cancer incidence risk to the amount of plutonium inhaled.

Calculation of the *TIC* with uncertainty was discussed in a previous section. Uncertainty in risk estimates includes uncertainty in the *TIC* and risk coefficients. Receptor behavior patterns

(i.e., the time spent doing different activities at different exertion levels) and their physical attributes (body weight and breathing rate) were considered fixed quantities.

Table 11. Time Budgets and Weighted Breathing Rates for the Exposure Scenarios

		Fraction of time spent at a given activity level					Weighted breathing rate
Scenario	Activity	Resting	Sitting	Light	Heavy	Hours	(m^3 h^{-1})
Rancher	Occupational	0.00	0.00	0.25	0.75	1.0	2.625
	Nonoccupational	0.00	0.50	0.38	0.13	7.0	1.208
	Sleeping	1.00	0.00	0.00	0.00	1.0	0.450
	Weighted Daily Average						1.281
Office Worker	Occupational	0.00	0.25	0.75	0.00	1.0	1.042
	Nonocuupational	0.00	0.50	0.38	0.13	7.0	1.004
	Sleeping	1.00	0.00	0.00	0.00	1.0	0.324
	Weighted Daily Average						0.932
Housewife	Occupational	0.00	0.13	0.75	0.13	1.0	1.331
	Indoor nonoccupational	0.00	0.50	0.38	0.13	3.0	1.004
	Outdoor nonoccupational	0.00	0.38	0.50	0.13	4.0	1.113
	Total Nonoccupational	0.00	0.44	0.44	0.13	7.0	1.066
	Sleeping	1.00	0.00	0.00	0.00	1.0	0.324
	Weighted Daily Average						1.013
Laborer	Occupational	0.00	0.125	0.50	0.375	8.0	1.943
	Indoor nonoccupational	0.00	0.5	0.375	0.125	1.0	1.208
	Outdoor nonoccupational	0.00	0.5	0.25	0.25	0.0	1.395
	Total nonoccupational	0.00	0.50	0.31	0.19	1.0	1.208
	Sleeping	1.00	0.00	0.00	0.00	0.0	0.450
	Weighted Daily Average						1.861
Infant	Awake—sedintary	0.00	0.71	0.14	0.14	3.0	0.334
	Awake—active	0.00	0.00	1.00	0.00	3.0	0.447
	Sleeping	1.00	0.00	0.00	0.00	3.0	0.144
	Weighted Daily Average						0.308
Child (2–6)	Indoor (home)	0.00	0.50	0.42	0.08	3.0	0.549
	Outdoor (home)	0.00	0.00	0.67	0.33	0.0	1.040
	Total Home					3.0	0.794
	Indoor (school)	0.00	0.80	0.20	0.00	0.0	0.351
	Sleeping	1.00	0.00	0.00	0.00	6.0	0.228
	Weighted Daily Average						0.417
Student (7–18)	Indoor home	0.00	0.44	0.56	0.00	3.0	0.829
	Outdoor home	0.00	0.00	0.25	0.75	3.0	1.979
	Total home	0.00	0.22	0.40	0.38	6	1.404
	Indoor school	0.00	0.75	0.25	0.00	0.0	0.591
	Outdoor school	0.00	0.00	0.25	0.75	0.0	1.979
	Total school	0.00	0.38	0.25	0.38	0	0.000
	Sleeping	1.00	0.00	0.00	0.00	3.0	0.326
	Weighted Daily Average						1.045

The amount of plutonium inhaled by a receptor for the 9-hour exposure period is given by

$$I = TIC \frac{\left(WBR_1\, T_1 + WBR_2\, T_2 + WBR_3\, T_3\right)}{ED} \quad (26)$$

where

I = intake of plutonium by the receptor for the exposure period (Ci)

TIC = time-integrated concentration (Ci-h m^{-3})

$WBR_{1,2,3}$ = time-weighted average breathing rate for occupational, nonoccupational, and sleeping activity (m^3 h^{-1})

$T_{1,2,3}$ = hours during the exposure period for occupational, nonoccupational, and sleeping activity (h)

ED = exposure duration (9 hours).

The subscripts 1, 2, and 3 refer to occupational, nonoccupational, and sleeping activity, respectively. Note that the values for *WBR* in Table 11 are weighted toward the time spent doing occupational and nonoccupational activity to account for exposure to the bulk of the release during waking hours.

Risk Coefficients

Calculating the lifetime cancer incidence risk requires estimates of risk coefficients. Risk coefficients relate the lifetime risk of cancer incidence to the amount of plutonium inhaled. Plutonium risk coefficients were developed in Phase II of the study and are documented in Grogan et al. (1999).

Plutonium emits alpha particles that have such weak penetration abilities that they can be blocked by a piece of paper or the dead, outer layers of the skin. As a result, the major danger from plutonium comes from having it inside the body. For residents in the vicinity of Rocky Flats, plutonium is most likely to have entered the body from breathing air that contained plutonium particles released from the site. After inhalation, plutonium enters the blood and about 80% is transported to the bone or liver where it is retained for years. Following inhalation, the four most highly exposed tissues are bone surface, lung, liver, and bone marrow. These tissues account for more than 97% of the total dose received by infants and adults alike. The dose per unit activity inhaled varies for these four tissues (Table 12). Furthermore, the dose per unit activity (dose conversion factor) also varies depending on the particle size distribution of the inhaled plutonium aerosol (Table 12). Three different particle size distributions are used to characterize the broad range of particles that may have been released during the 1957 fire: 1-, 5-, and 10-µm activity median aerodynamic diameter (AMAD) particles. Each of these distributions is assumed to be lognormal with a GSD of 2.5; therefore, each size distribution covers a relatively large range of particle sizes. The 1-µm AMAD particle size distribution results in the largest doses to the tissues per unit intake of activity because the particles penetrate deeper into the lungs and are retained longer. In all cases, the plutonium is assumed to be in the oxide form.

Table 12. Summary of Plutonium Oxide Inhalation Dose Conversion Factors[a]

	Dose conversion factor (μGy Bq^{-1}) [a]		
Cancer site	1-μm AMAD particles GSD = 2.5	5-μm AMAD particles GSD = 2.5	10-μm AMAD particles GSD = 2.5
Lung	4.4 (1.9)	2.6 (2.7)	1.2 (4.3)
Liver	2.0 (3.0)	0.95 (3.5)	0.42 (4.5)
Bone surface	9.0 (3.0)	4.6 (3.5)	2.1 (4.5)
Bone marrow	0.46 (3.0)	0.22 (3.5)	0.11 (4.5)

[a.] Values for 1-μm AMAD from ICRP 1995; 5 and 10-μm were calculated in Grogan et al. 1999
[b.] Geometric mean (geometric standard deviation).

The incidence of health effects depends on the amount of dose received. There are two main classes of health effects induced by ionizing radiation: deterministic and stochastic effects. Deterministic effects most often follow acute, high dose exposure. The severity of the effect increases with dose above the threshold dose. Below the threshold dose, the effect is not evident; however, subtle minor effects may occur. Deterministic effects cause direct damage to tissues and include effects that most often occur within days to weeks after exposure. For example, these effects can cause reddening of the skin, cataracts, hair loss, sterility, and bone marrow depression after external irradiation. After inhalation of plutonium, deterministic effects may include radiation pneumonitis, pulmonary fibrosis, and lymphopenia, but these conditions occur only after very high doses. The threshold dose for most deterministic effects is at least 0.5 Gy delivered in a short time, and many are much higher (NCRP 1991). For the releases of plutonium that occurred from the site, doses to individuals in the Rocky Flats area were well below the threshold doses. Therefore, deterministic health effects were not possible.

Stochastic effects are assumed to occur randomly at all dose levels, including the lowest doses. The frequency of stochastic effects is dependent on the dose, and the effects usually occur at long intervals after exposure. In a large population exposed to low doses, only a few of the exposed individuals will be affected, most will not. The two principal types of stochastic effects are induced cancer and genetic effects. For exposure to plutonium, the risk of induced cancer is the health effect of most concern; in particular, lung cancer, liver cancer, bone cancer, and leukemia (bone marrow exposure) because these are the tissues that receive the highest doses. Genetic effects are not an important risk for plutonium exposures because (1) people exposed to radiation are several times more likely to be affected by an induced cancer than to transmit genetic effects to their children and (2) the plutonium doses to the gonads (ovaries or testes) are small compared to other organs of the body (40 times less than the lung). Therefore, we did not consider them further.

The alpha particles emitted from plutonium are densely ionizing, and the linear energy transfer (LET) to the tissue is high over the short range (about 40 μm) of the alpha particles (thus, the name high-LET radiation). Other radiations, such as gamma rays and x-rays, are less densely ionizing and are termed low-LET radiations. The biological effects of low-LET radiation are better known than those of high-LET radiation. The differences between radiation types are important to the analysis because high-LET radiations are more biologically effective (cause more damage) per unit of dose than low-LET radiations. This difference in effectiveness is usually described by the relative biological effectiveness (RBE), which is the ratio of doses from two different radiations to produce the same type and level of biological effect.

Inhalation of plutonium results in the exposure of organs to high-LET radiation. While a few human populations have been exposed directly to large amounts of plutonium and some populations to other radionuclides that emit alpha particles, more groups have been exposed to low-LET gamma radiation and evaluated in more epidemiologic detail. In addition, studies of cancer in animals exposed to both types of radiation and laboratory studies of cellular and other biological endpoints can be used to support human studies. These different sources of information were used in this phase of the study to develop four independent approaches to estimate the risk of cancer because of radiation doses from plutonium deposited in the organs of the human body (Grogan et al. 1999). Three approaches used epidemiologic studies of human populations to derive dose-response relationships, and the fourth used dose-response relationships from controlled animal experiments. The four independent approaches were used to derive, where possible, risk coefficients for each organ of interest. The coefficients from the different approaches were then combined by weighting each according its intrinsic merit to produce a single risk coefficient with uncertainties for each organ of interest.

The overall mortality risk estimate for each cancer site was adjusted by the lethality fraction to provide lifetime risk estimates for cancer incidence. The influence of gender and age was accounted for in the analyses (see Grogan et al. [1999] for details). The data allowed a distinction to be made between the risks and uncertainties to those under 20 years of age at exposure and those 20 and older. The data did not warrant a more detailed analysis. For this reason, the risk coefficients for persons under 20 years of age were applied to the infants and children in the seven hypothetical exposure scenarios.

The GM (50th percentile) and GSDs of the cancer incidence risk coefficient distributions are listed in Table 13. The units reported in Grogan et al. (1999) have been changed from risk per 100,000 persons per unit of activity in kilobecquerels (kBq) to risk per 10,000 persons per unit of activity in microcuries (µCi). These numbers indicate the median number of cases of cancer (fatal and nonfatal) that would be expected to result from 10,000 people all inhaling 1 µCi of $^{239/240}$Pu particles with the defined particle size distribution.

Because the particle size distribution of the effluent is unknown, other than it is all respirable, we have treated it stochastically. The particle size used in each RATCHET Monte Carlo trial was passed to the dose calculation, thereby, allowing the appropriate risk coefficients to be selected based on the particle size. We used the following cutoffs in the calculation:

- If the particle size <2.5 µm AED, then the 1-µm AMAD risk coefficients were used
- If the particle size was between 2.5 and 7.5 µm AED, then the 5-µm AMAD risk coefficients were used
- If the particle size was >7.5 µm AED, then the 10-µm AMAD risk coefficients were used.

This procedure allowed the particle size used in the transport simulation to be correlated to the particle-size specific risk coefficient.

Table 13. Lifetime Cancer Incidence Risk Per 10,000 Persons Per 1 µCi of Inhaled Plutonium for the Three Particle Size Distributions Used to Characterize the 1957 Fire Releases[a]

1-µm AMAD particles (GSD = 2.5)			
Cancer site	Gender	Under 20	20 and older
Lung	Male	206 (3.5)	210 (3.4)
	Female	206 (3.5)	210 (3.4)
Liver	Male	92 (5.2)	49 (5.2)
	Female	45 (5.4)	23 (5.4)
Bone surface	Male	16 (9.5)	8.0 (9.3)
	Female	8.0 (10)	4.0 (10)
Bone marrow	Male	2.4 (6.1)	2.3 (6.3)
	Female	2.4 (6.1)	2.3 (6.3)
5-µm AMAD particles (GSD = 2.5)			
Cancer site	Gender	Under 20	20 and older
Lung	Male	117 (4.3)	119 (4.2)
	Female	117 (4.3)	119 (4.2)
Liver	Male	46 (5.8)	24 (5.7)
	Female	21 (6.0)	11 (6.0)
Bone surface	Male	8.3 (10)	4.3 (10)
	Female	4.1 (11)	2.1 (11)
Bone marrow	Male	1.1 (6.7)	1.1 (6.7)
	Female	1.1 (6.7)	1.1 (6.7)
10-µm AMAD particles (GSD = 2.5)			
Cancer site	Gender	Under 20	20 and older
Lung	Male	55 (6.1)	56 (6.0)
	Female	55 (6.1)	56 (6.0)
Liver	Male	21 (6.7)	11 (6.8)
	Female	9.6 (7.0)	5.0 (6.9)
Bone surface	Male	4.0 (12)	2.1 (12)
	Female	2.0 (12)	1.0 (12)
Bone marrow	Male	0.54 (7.9)	0.54 (8.0)
	Female	0.54 (7.9)	0.49 (8.0)

[a] Geometric mean (geometric standard deviation).

Risk Calculations

Plutonium intake (Equation [26]) was multiplied by the risk coefficients (Table 13) and summed across all particle sizes to yield lifetime cancer incidence risk (R_j) for each organ of interest.

$$R_j = I \times RC_j \tag{27}$$

where

R_j = lifetime cancer incidence risk for the j^{th} organ

I = plutonium intake (Ci)

RC_j = risk coefficient for the j^{th} organ (Ci^{-1}).

Recall that the risk coefficient and plutonium intake are correlated by particle size. Risk coefficients for the different organs were also assumed to be correlated. For each Monte Carlo trial, a standard normal deviate was generated and stored. These deviates were then used to determine a risk coefficient for each organ of interest using Equation (28), substituting the appropriate GM and GSD of the specific organ.

$$RC = \exp\left(d \ln(GSD) + \ln(GM)\right) \tag{28}$$

The total lifetime cancer incidence risk from all organs was calculated by summing the risk across all four organs during each Monte Carlo trial.

Risk calculations were performed using a FORTRAN programs that (1) read *TIC* values from the RATCHET output file, (2) computed plutonium intake, (3) sampled risk coefficients and calculated risk, and (4) stored and processed output. FORTRAN routines for generating random numbers and normal deviates were adapted from Press et al. (1992). The output distributions provided in this report were generated from 1000 trials.

LIFETIME CANCER INCIDENCE RISK ESTIMATES

Incremental lifetime cancer incidence risks were expressed in terms of percentiles of the cumulative density function (Tables 14–16). Lognormal statistics (that is, GM and GSD) were not possible because for many of the receptors the risk was zero for some fraction of the Monte Carlo trials. The value of the incremental lifetime cancer incidence risk depended not only on the location of the receptor but the percentile value chosen. At the 2.5% level, the receptor with the maximum total (all organs) risk in the model domain was the laborer located east of the city of Arvada (5.8×10^{-7}) followed by the laborer located at Federal Heights (5.4×10^{-7}). At the 50% level, the receptor with the maximum risk was also the laborer located east of Arvada, but the person with the second highest risk was the rancher located south of the RFP buffer zone. At the 97.5% level, the rancher located south of the RFP buffer zone had the highest risk (1.7×10^{-4}), followed by the rancher located west of Arvada (3.4×10^{-5}). The 95% uncertainty range (from 2.5 to 97.5%) for the rancher located south of the RFP buffer was from 0 to 1.7×10^{-4}. Using the laborer located east of Arvada as an example, the uncertainty in these risk estimates may be interpreted as follows:

- There is a 95% probability that the incremental lifetime cancer incidence risk was between 5.8×10^{-7} (2.5% value) and 1.6×10^{-5} (97.5% value).
- There is a 2.5% probability that the incremental lifetime cancer incidence risk was between 1.6×10^{-5} and 6.9×10^{-5} (100% value, see Appendix B) and a 2.5% probability the risk was between 5.2×10^{-8} (0% value, see Appendix B) and 5.8×10^{-7}.

We may also interpret this to mean, given an exposure history and lifestyle similar to that of the laborer, there is a 97.5% probability that the model predicted number of cancer cases attributed to inhalation of plutonium originating from the 1957 fire release would be no greater than 16 persons in a population of 1 million similarly exposed individuals. In all cases, the organ with the greatest risk was the lung, followed by the liver, bone surface, and bone marrow.

Receptors located in the cities of Arvada, Westminster, Northglenn, Thornton, and Federal Heights had the highest probability of being exposed (see Figure 8). In the 1000 RATCHET Monte Carlo trials, the plume never reached the city of Boulder; therefore, the risk was zero at

that location. While the rancher located south of the RFP buffer zone had the highest risk at the 97.5% level, the probability of him being exposed was only about 60%—that is, only 60% of the RATCHET simulations predicted the plume to reach that location (see Figure 8). This added level of uncertainty makes it difficult to interpret results in the same fashion as those for routine releases (Rood 1999b).

An almost infinite number of possible exposure scenarios can be defined; in most cases, the risks associated with each scenario will differ. However, the maximum risks will probably be bounded by the risks associated with the rancher and laborer scenarios. These scenarios may be considered to represent the maximum exposed individuals in the model domain because they were placed at the point of highest concentration outside the RFP buffer zone. In addition, the laborer was assumed to be working a graveyard shift, thereby, maximizing his breathing rate during the releases from the fire.

The median (50%) risk estimates calculated for Phase II are similar to those calculated in Phase I. Geometric mean risk for Phase I, within 3 miles (4.8 km) of the RFP was around 1×10^{-6} while the 50% risk values for Phase II in a similar location were about 2×10^{-6}. However, Phase I and II airborne activity concentration estimates were substantially different from one another. The maximum 13.5-hour average concentration in Phase I was around 0.62 pCi m^{-3}. For Phase II, the maximum 9-hour average concentration at the 50% level was around 6 pCi m^{-3}. The higher concentrations calculated in Phase II was primarily due to the substantially higher source term used. It is coincidental that the final cancer incidence risk estimates are about the same in both Phases of the study. Airborne concentrations in Phase II were higher than Phase I, therefore, compensating differences also existed in the calculation of dose and risk between the two Phases.

Phase I determined preliminary risk estimates and therefore used a less rigorous approach than in Phase II in estimating radiation dose and cancer incidence risk. In Phase I the plutonium inhalation dose conversion factor was represented by a uniform distribution between the values given in ICRP Report 56 (ICRP 1990) for slightly soluble (Class W) and insoluble (Class Y) forms of plutonium. The dose conversion factor ranged from 1.2×10^{-4} Sv Bq^{-1} to 8.4×10^{-5} Sv Bq^{-1}. The plutonium particle size distribution was assumed to be lognormal with an AMAD of 1 µm. In Phase II, uncertainty in the inhalation dose conversion factor for a broad range of particle size distributions was explicitly accounted for (1 µm AMAD, 5 µm AMAD, 10 µm AMAD).The dose conversion factors are smaller for the larger size distributions. This resulted in significantly different values for the conversion from intake to risk compared to Phase I. In Phase II, the plutonium released during the fire was postulated to be plutonium oxide and highly insoluble. Lifetime cancer incidence risk coefficients with uncertainties for exposure to plutonium were estimated for the principle organs of concern: lung, liver, bone and bone marrow. In contrast, a single value for the whole body risk estimate of 7.3 percent per sievert (ICRP 1990) was assumed in Phase I.

Table 14. Lifetime Incremental Cancer Incidence Risk at the 2.5% level for 1957 Fire Releases

Receptor	Location	Lung	Liver	Bone surface	Bone marrow	Total
Laborer	Arvada East	4.5×10^{-7}	1.1×10^{-7}	2.2×10^{-8}	4.7×10^{-9}	5.8×10^{-7}
Laborer	Federal Heights	4.2×10^{-7}	9.8×10^{-8}	2.1×10^{-8}	4.4×10^{-9}	5.4×10^{-7}
Rancher	Arvada West	---[a]	---[a]	---[a]	---[a]	---[a]
Rancher	So RFP Buffer	---[a]	---[a]	---[a]	---[a]	---[a]
Housewife	Arvada	2.2×10^{-8}	2.4×10^{-9}	5.2×10^{-10}	2.3×10^{-10}	2.5×10^{-8}
Housewife	Wheat Ridge	---[a]	---[a]	---[a]	---[a]	---[a]
Housewife	Westminster	1.0×10^{-8}	1.1×10^{-9}	2.4×10^{-10}	1.0×10^{-10}	1.1×10^{-8}
Housewife	Broomfield	---[a]	---[a]	---[a]	---[a]	---[a]
Laborer	Thornton	2.1×10^{-7}	5.0×10^{-8}	1.0×10^{-8}	2.2×10^{-9}	2.7×10^{-7}
Infant	Northglenn	5.7×10^{-8}	1.2×10^{-8}	2.7×10^{-9}	6.0×10^{-10}	7.2×10^{-8}
Laborer	Boulder	---[a]	---[a]	---[a]	---[a]	---[a]
Laborer	Lafayette	---[a]	---[a]	---[a]	---[a]	---[a]
Student	Golden	---[a]	---[a]	---[a]	---[a]	---[a]
Child	Commerce City	5.0×10^{-12}	1.1×10^{-12}	2.4×10^{-13}	5.3×10^{-14}	6.4×10^{-12}
Office Worker	Denver	---[a]	---[a]	---[a]	---[a]	---[a]

a. The modeled air concentration at this percentile level was zero

Table 15. Lifetime Incremental Cancer Incidence Risk at the 50% level for 1957 Fire Releases

Receptor	Location	Lung	Liver	Bone surface	Bone marrow	Total
Laborer	Arvada East	2.2×10^{-6}	5.2×10^{-7}	1.1×10^{-7}	2.3×10^{-8}	2.9×10^{-6}
Laborer	Federal Heights	1.9×10^{-6}	4.5×10^{-7}	9.4×10^{-8}	2.0×10^{-8}	2.5×10^{-6}
Rancher	Arvada West	1.9×10^{-6}	4.5×10^{-7}	9.5×10^{-8}	2.0×10^{-8}	2.5×10^{-6}
Rancher	So RFP Buffer	2.2×10^{-8}	5.1×10^{-9}	1.1×10^{-9}	2.3×10^{-10}	2.8×10^{-8}
Housewife	Arvada	4.7×10^{-7}	5.2×10^{-8}	1.1×10^{-8}	4.9×10^{-9}	5.3×10^{-7}
Housewife	Wheat Ridge	1.4×10^{-7}	1.6×10^{-8}	3.4×10^{-9}	1.5×10^{-9}	1.6×10^{-7}
Housewife	Westminster	2.1×10^{-7}	2.3×10^{-8}	5.0×10^{-9}	2.2×10^{-9}	2.4×10^{-7}
Housewife	Broomfield	8.9×10^{-12}	9.9×10^{-13}	2.1×10^{-13}	9.3×10^{-14}	1.0×10^{-11}
Laborer	Thornton	1.1×10^{-6}	2.6×10^{-7}	5.5×10^{-8}	1.2×10^{-8}	1.4×10^{-6}
Infant	Northglenn	2.6×10^{-7}	5.7×10^{-8}	1.3×10^{-8}	2.8×10^{-9}	3.4×10^{-7}
Laborer	Boulder	---[a]	---[a]	---[a]	---[a]	---[a]
Laborer	Lafayette	---[a]	---[a]	---[a]	---[a]	---[a]
Student	Golden	---[a]	---[a]	---[a]	---[a]	---[a]
Child	Commerce City	4.0×10^{-8}	8.7×10^{-9}	1.9×10^{-9}	4.2×10^{-10}	5.1×10^{-8}
Office Worker	Denver	7.1×10^{-9}	7.9×10^{-10}	1.7×10^{-10}	7.4×10^{-11}	8.2×10^{-9}

a. The modeled air concentration at this percentile level was zero

Table 16. Lifetime Incremental Cancer Incidence Risk at the 97.5% level for 57 Fire Releases

Receptor	Location	Lung	Liver	Bone surface	Bone marrow	Total
Laborer	Arvada East	1.3×10^{-5}	2.9×10^{-6}	6.2×10^{-7}	1.3×10^{-7}	1.6×10^{-5}
Laborer	Federal Heights	8.8×10^{-6}	2.1×10^{-6}	4.3×10^{-7}	9.1×10^{-8}	1.1×10^{-5}
Rancher	Arvada West	2.6×10^{-5}	6.1×10^{-6}	1.3×10^{-6}	2.7×10^{-7}	3.4×10^{-5}
Rancher	So RFP Buffer	1.3×10^{-4}	3.1×10^{-5}	6.5×10^{-6}	1.4×10^{-6}	1.7×10^{-4}
Housewife	Arvada	1.4×10^{-5}	1.6×10^{-6}	3.4×10^{-7}	1.5×10^{-7}	1.6×10^{-5}
Housewife	Wheat Ridge	5.5×10^{-6}	6.1×10^{-7}	1.3×10^{-7}	5.8×10^{-8}	6.3×10^{-6}
Housewife	Westminster	2.2×10^{-6}	2.5×10^{-7}	5.4×10^{-8}	2.3×10^{-8}	2.6×10^{-6}
Housewife	Broomfield	2.3×10^{-8}	2.5×10^{-9}	5.5×10^{-10}	2.4×10^{-10}	2.6×10^{-8}
Laborer	Thornton	4.5×10^{-6}	1.1×10^{-6}	2.2×10^{-7}	4.7×10^{-8}	5.9×10^{-6}
Infant	Northglenn	1.2×10^{-6}	2.6×10^{-7}	5.9×10^{-8}	1.3×10^{-8}	1.6×10^{-6}
Laborer	Boulder	---[a]	---[a]	---[a]	---[a]	---[a]
Laborer	Lafayette	5.7×10^{-10}	1.3×10^{-10}	2.8×10^{-11}	6.0×10^{-12}	7.4×10^{-10}
Student	Golden	2.2×10^{-6}	4.8×10^{-7}	1.1×10^{-7}	2.4×10^{-8}	2.8×10^{-6}
Child	Commerce City	4.0×10^{-7}	8.6×10^{-8}	1.9×10^{-8}	4.2×10^{-9}	5.1×10^{-7}
Office Worker	Denver	3.7×10^{-7}	4.1×10^{-8}	9.0×10^{-9}	3.9×10^{-9}	4.3×10^{-7}

[a] The modeled air concentration at this percentile level was zero

REFERENCES

Briggs, G.A. 1969. *Plume Rise*. TID-25075. U.S. Atomic Energy Commission, Washington, D.C.

Briggs, G.A. 1975. "Plume Rise Predictions." *Lectures on Air Pollution and Environmental Impact Analysis.* American Meteorological Society, Boston, Massachusetts. 59–111.

Briggs, G.A. 1984. "Plume Rise and Buoyancy Effects." *Atmospheric Science and Power Production.* DOE/TIC-27601. U.S. Department of Energy, Washington, D.C. 327–366.

Brown, K.J. 1991. *Rocky Flats 1990-91 Winter Validation Tracer Study*. Report AG91-19, North American Weather Consultants, Salt Lake City, Utah.

Byun, D.W. and R. Dennis 1995. "Design Artifacts in Eulerian Air Quality Models: Evaluation of the Effects of Layer Thickness and Vertical Profile Correction on Surface Ozone Concentrations." *Atmospheric Environment*, (29): 105–126.

ChemRisk. 1994a. *Estimating Historical Emissions from Rocky Flats 1952-1989*. Project Task 5 for Phase I. Prepared for the Colorado Department of Public Health and Environment. ChemRisk, A division of McLaren/Hart, 1135 Atlantic Avenue, Alameda, California 94501. March.

ChemRisk. 1994b. *Reconstruction of Historical Rocky Flats Operations & Identification of Release Points*. Project Task 3&4 for Phase I. Prepared for the Colorado Department of Public Health and Environment. ChemRisk, A division of McLaren/Hart, 1135 Atlantic Avenue, Alameda, California 94501. May

ChemRisk. 1994c. *Exposure Pathway Identification and Transport Modeling*. Project Task 6 for Phase I. Prepared for the Colorado Department of Public Health and Environment. ChemRisk, A division of McLaren/Hart, 1135 Atlantic Avenue, Alameda, California 94501. May.

ChemRisk. 1994d. *Dose Assessment for Historical Contaminant Releases from Rocky Flats*. Project Task 8 for Phase I. Prepared for the Colorado Department of Public Health and Environment. ChemRisk, A division of McLaren/Hart, 1135 Atlantic Avenue, Alameda, California 94501. September.

ChemRisk 1994e. *Demographic & Land Use Reconstruction of the Area Surrounding the Rocky Flats Plant*. Project Task 7 for Phase I. Prepared for the Colorado Department of Public Health and Environment. ChemRisk, A division of McLaren/Hart, 1135 Atlantic Avenue, Alameda, California 94501. April.

Crow, L.W. 1974. *Characteristic Airflow Patterns Near Rocky Flats Plant and Their Relationship to Metropolitan Denver*. LWC-143. Report prepared for Dow Chemical USA, Rocky Flats Division. December.

DOE (U.S. Department of Energy). 1980. *Final Environmental Impact Statement, Rocky Flats Plant*. DOE/EIS-0064. April.

DOE (U.S. Department of Energy). 1995. *Rocky Flats Environmental Technology Site Historical Data Summary*. AV-R-93-08-200. February.

Elderkin, C. E. and P. H. Gudiksen. 1993. "Transport and Dispersion in Complex Terrain." *Radiation Protection Dosimetry* 50: 265–271.

EPA. 1987. *On-Site Meteorological Program Guidance for Regulatory Modeling Applications*. EPA-450/4–87–013. Research Triangle Park, North Carolina.

EPA. 1992. *User's Guide for the Industrial Source Complex (ISC) Dispersion Models Vol. 1, User's Instructions*. EPA-450/4–92–008a. Research Triangle Park, North Carolina.

Epp, J.G., W.R. Cornelison, R.R. Harrison, B.P. Shepherd, E.R. Turnquist, and H.A. Spavin. 1957a. *Report of Investigation of Serious Incident in Building 71 on September 11, 1957* (unclassified version). Dow Chemical Company. Repository Document IN-732. October 7.

Epp, J.G., W.R. Cornelison, R.R. Harrison, B.P. Shepherd, E.R. Turnquist, and H.A. Spavin. 1957b. *Supplementary Report on Fire in Building 71 on September 11, 1957*. Dow Chemical Company. Repository Document IN-756. December 10.

Gifford, F.A. 1961. "Use of Routine Meteorological Observations for Estimating Atmospheric Dispersion." *Nuclear Safety* 2 (4): 47–51.

Gifford, F.A. 1983. "Atmospheric Diffusion in the Mesoscale Range: Evidence of Recent Plume Width Observations." *Sixth Symposium on Turbulence and Diffusion, American Meteorological Society*. Boston, Massachusetts.

Golder, D. 1972. "Relations among Stability Parameters in the Surface Layer." *Boundary-Layer Meteorology* 3 (1): 47–58.

Grogan, H.A., W.K. Sinclair, and P.G. Voillequé. 1999. *Assessing Risks of Exposure to Plutonium*. *RAC* Report No. 5-CDPHE-RFP-1998-FINAL(Rev. 1). *Radiological Assessments Corporation*, Neeses, South Carolina. August

HAP (Health Advisory Panel). 1993. *Health Advisory Panel's Report to Colorado Citizens on the Phase I Study of the State of Colorado's Health Studies on Rocky Flats*. Colorado Department of Public Health and Environment, Denver, Colorado.

Hammond, S.E. 1957. *Monthly Progress Report Site Survey–October 1957*. The Dow Chemical Company, Rocky Flats Division, Denver, Colorado. November 6.

Hammond, S.E. 1958. *Monthly Progress Report Site Survey–May 1958.* The Dow Chemical Company, Rocky Flats Division, Denver, Colorado. June 5.

Hanna, S.R., G.A. Briggs, and R.P. Hosker, Jr. 1982. *Handbook on Atmospheric Diffusion*. DOE/TIC-11223. U.S. Department of Energy, Office of Health and Environmental Research.

Hatch, W.L. 1988. *Selective Guide to Climatic Data Sources*. Key to Meteorological Records Documentation No. 4.11. National Climatic Data Center, Asheville, North Carolina.

Harper, F.T., S.C. Hora, M.L. Young, L.A. Miller, C.H. Lui, M.D. McKay, J.C. Helton, L.H.J. Goossens, R.M. Cooke, J. Pasler-Sauer, B. Kraan, and J.A. Jones. 1995. *Probability Accident Consequence Uncertainty Analysis, Dispersion and Deposition Uncertainty Assessment*. NUREG/CR-6244. U.S. Nuclear Regulatory Commission, Washington, D.C.

Hicks, B.B., K.S. Rao, R.J. Dobosy, R.P. Hosker, J.A. Herwehe, and W.R. Pendergrass. 1989. *TRIAD: A Puff-Trajectory Model for Reactive Gas Dispersion with Application to UF_6 Releases to the Atmosphere.* ERL ARL-168. National Oceanic and Atmospheric Administration, Air Resources Laboratory, Silver Springs, Maryland.

Hodgin, C.R. 1991. *Terrain-Responsive Atmospheric Code (TRAC) Transport and Diffusion: Features and Software Overview*. Report RFP-4516. EG&G Rocky Flats, Golden, Colorado.

ICRP (International Commission on Radiological Protection). 1990. *1990 Recommendations of the International Commission on Radiological Protection.* ICRP Publication 60. *Annals of the ICRP* 21, Nos 1–3. Pergamon Press, Oxford.

ICRP 1995. *Age-Dependent Doses to Members of the Public from Intake of Radionuclides: Part 4, Inhalation Dose Coefficients*. ICRP Publication 71. Ann. ICRP 25, Nos 3 & 4. Pergamon Press, Oxford.

Kathren, R.L. 1974. "Towards Interim Report BNWL-SA-1510." In J.B. Owen, L.M. Steward, *A Historical Summation of Environmental Incidents Affecting Soils at or Near the USAEC Rocky Flats Plant*. Repository Document RE-318.

Layton, D.W. 1993. "Metabolically Consistent Breathing Rates for Use In Dose Assessment." *Health Physics* 64 (1): 23–36.

Luna, R.E. and H.W. Church. 1972. "A Comparison of Turbulence Intensity and Stability Ratio Measurements to Pasquill Stability Classes." *Journal of Applied Meteorology* 11 (4): 663–669.

Martell, E. 1975. *Actinides in the Environment and Their Uptake by Man*. National Center for Atmospheric Research Technical Note NCAR-TN/STR-110. Repository Document IP-807. May.

NCRP (National Council on Radiological Protection and Measurement). 1991. *Some Aspects of Strontium Radiobiology*. NCRP Report No. 110. National Council on Radiological Protection and Measurement, Bethesda, Maryland.

Panofsky, H.A. and J.A. Dutton. 1984. *Atmospheric Turbulence*. New York: J. Wiley and Sons.

Pasquill, F. 1961. "The Estimation of the Dispersion of Windborne Material." *The Meteorological Magazine* 90: 33–49.

Pasquill, F. and F.B. Smith. 1983. *Atmospheric Diffusion*, 3rd Edition. New York: Halsted Press.

Petersen, W.B. and L.G. Lavdas. 1986. *INPUFF 2.0 - A Multiple Source Gaussian Puff Dispersion Algorithm: User's Guide*. EPA-600/8-86/024. Atmospheric Sciences Research Laboratory, U.S. Environmental Protection Agency, Research Triangle Park, North Carolina.

Pleim, J., A. Venkatram, and R Yamartino, 1984. *ADOM/TADAP Model Development Program. Volume 4, The Dry Deposition Module*. Ontario Ministry of Environment, Rexdale, Ontario.

Press, W.H., S.A. Teukolsky, W.T. Vetterling, and B.P. Flannery. 1992. *Numerical Recipes: The Art of Scientific Computing*. New York: New York: Cambridge University Press.

Ramsdell, J.V. Jr., C.A. Simonen, and K.W. Burk. 1994. *Regional Atmospheric Transport Code for Hanford Emission Tracking* (RATCHET). PNWD-2224 HEDR. Battelle Pacific Northwest Laboratories, Richland, Washington.

Rope, S.K., K.R. Meyer, M.J. Case, H.A. Grogan, D.W. Schmidt, M. Dreicer, T.F. Winsor, and J.E. Till. 1999. *Task 4: Evaluation of Environmental Data for Historical Public Exposure Studies on Rocky Flats*. *RAC* Report No. 1-CDPHE-RFP-1997-Final (Rev. 1). *Radiological Assessments Corporation,* Neeses, South Carolina. August

Rood, A.S. 1999a. *Rocky Flats Dose Reconstruction Project, Phase II: Performance Evaluation of Atmospheric Transport Models*. *RAC* Report No. 3-CDPHE-RFP-1996-FINAL (Rev.1). *Radiological Assessments Corporation,* Neeses, South Carolina. August.

Rood, A.S. 1999b. *Estimated Exposure and Lifetime Cancer Incidence Risk from Routine Plutonium Releases at the Rocky Flats Plant. RAC* Report No. 8-CDPHE-RFP-1997-FINAL (Rev.1). *Radiological Assessments Corporation*, Neeses, South Carolina. August.

Roy, M. and C. Courtay. 1991. "Daily Activities and Breathing Parameters for Use in Respiratory Tract Dosimetry." *Radiation Protection Dosimetry* 35 (3): 179–186.

Schere, K.L. and C.J. Coates. 1992. "A Stochastic Methodology for Regional Wind-Field Modeling." *Journal of Applied Meteorology* 31 (12): 1407–1425.

Seinfeld, J.H. 1986. *Atmospheric Chemistry and Physics of Air Pollution*. New York: John Wiley and Sons.

Slinn, S.A. and W.G.N. Slinn, 1980. "Predictions for Particle Deposition in Natural Waters." *Atmospheric Environment* (14): 1031–1-16.

Slinn, W.G.N., 1982. "Predictions for Particle Deposition to Vegetative Canopies." *Atmospheric Environment* (16): 1785–1974.

Stull, R.B. 1988. *An Introduction to Boundary Layer Meteorology*. Dordrecht, Netherlands: Kluwer Academic Publishers.

Turner, D.B. 1964. "A Diffusion Model for an Urban Area." *Journal of Applied Meteorology* 3 (1): 83–91.

Voillequé, P.G. 1999a. *Estimated Airborne Releases of Plutonium During the 1957 Fire in Building 71*. RAC Report No. 10-CDPHE-RFP-1999-FINAL. *Radiological Assessments Corporation*, Neeses, South Carolina. August.

Voillequé, P.G. 1999b. *Review of Routine Releases of Plutonium in Airborne Effluents at Rocky Flats. RAC* Report No. 6-CDPHE-RFP-1998-FINAL. *Radiological Assessments Corporation,* Neeses, South Carolina. August

Weil, J.C. 1985. "Updating Applied Diffusion Models." *Journal of Climate and Applied Meteorology* 24 (11): 1111–1130.

Whicker, F.W. and V. Schultz. 1982. *Radioecology: Nuclear Energy and the Environment*. Boca Raton, Florida: CRC Press Inc.

Zilitinkevich, S.S. 1972. "On the Determination of the Height of the Ekman Boundary Layer." *Boundary-Layer Meteorology* 3 (2): 141–145.